



**UNITED STATES DEPARTMENT OF COMMERCE**  
**National Oceanic and Atmospheric Administration**  
National Marine Fisheries Service  
Southwest Fisheries Science Center  
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Santa Cruz, CA 95060

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Dear Interested Party:

The Technical Recovery Team (TRT) for the Oregon and Northern California Coast Recovery Domain has recently completed a draft report titled "Framework for assessing viability of threatened coho salmon in the Southern Oregon / Northern California Coast Evolutionarily Significant Unit." The purpose of this announcement is to seek comment from interested parties regarding the scientific content and analysis that underlie the recommendations and conclusions reached in this report.

Biological viability criteria specify those biological conditions that, if met, would indicate that populations and Evolutionarily Significant Units (ESUs) are at low risk of extinction. For the Southern Oregon / Northern California Coast (SONCC) Coho Salmon ESU, that includes coho salmon populations from Elk River (Oregon) in the north to Mattole River (California) in the south. Ideally, viability criteria for salmonid populations would be tailored to each population, accounting for specific biological characteristics for populations and differences in the inherent productive capacities of the habitats that underlie those attributes. However, the data required for such population-specific analyses are lacking throughout the entire ESU. Therefore we have developed a framework based on population-level criteria that are general to salmonids and specific to coho salmon when possible, with the expectation that these criteria can provide preliminary guidance on targets for viability, but may eventually be replaced with population-specific criteria as more information becomes available. Proposed ESU-level criteria are similarly derived from fundamental principles of conservation biology, providing guidance on configuration of viable populations (the units that constitute an ESU) that lead to a high likelihood of ESU persistence by ensuring representation of diversity within an ESU across most of its historical range, redundancy of viable populations to guard against larger-scale catastrophic risk, and ensuring connectivity among populations in order to maintain long-term ecological and evolutionary processes. We emphasize that the conclusions in this document do not represent policy decisions or include specific outcomes necessary for eventual delisting of the SONCC Coho Salmon ESU. In the process of preparing this report the TRT was forced to confront and accommodate a high degree of uncertainty in the analyses.

We invite comment on the scientific content and analysis present in this report. The report is available in electronic form at the website of the Fisheries Ecology Division of the NMFS Southwest Fisheries Science Center (<http://swfsc.noaa.gov/textblock.aspx?Division=FED&id=2268>). Comments should be sent electronically via email to Tommy Williams ([Tommy.Williams@noaa.gov](mailto:Tommy.Williams@noaa.gov)) with "Draft SONCC Viability Comments" in the subject line. We ask that comments be submitted by 31 July 2007.

Sincerely,

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**Framework for assessing viability of threatened coho salmon in the Southern Oregon / Northern California Coast Evolutionarily Significant Unit**

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## Preface

This report describes a framework for assessing coho salmon population viability that includes developing objective, measurable criteria that when met, would define when the Southern Oregon/Northern California Coast Coho Salmon Evolutionarily Significant Unit (ESU) is naturally self-sustaining with a low risk of extinction. Technical recovery planning for Pacific salmon and steelhead is intended to produce biologically based viability criteria for listed ESUs that will be considered in setting recovery goals.

The listing unit for Pacific salmon and steelhead is the ESU. ESUs are defined as a population or group of populations that are substantially reproductively isolated from other conspecific population units and that represent an important part of the evolutionary legacy of the species. The Southern Oregon/Northern California Coast (SONCC) Coho Salmon ESU includes coho salmon populations from Elk River (Oregon) in the north to Mattole River (California) in the south. This report provides a framework to assess the viability of individual populations within this region, and describes the spatial configuration of viable independent populations and dependent populations that would lead to a high likelihood of long-term ESU persistence.

This report constitutes a technical recommendation by the TRT intended to assist recovery planners in developing recovery strategies and prioritizing recovery actions. It does not constitute official agency policy. The Oregon and Northern California Coast TRT seeks comments from interested parties regarding the scientific content and analysis that underlie the recommendations and conclusions reached in this report. As a review draft, this document should not be cited; the report will be published in an appropriate venue when completed.

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## Abstract

*(Note: to be added in final draft)*

## Acknowledgments

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## 1. Introduction

The Endangered Species Act (ESA) requires that the National Marine Fisheries Service develop recovery plans for anadromous species listed as threatened or endangered and that these recovery plans contain objective, measurable criteria that when met, would define when a species would be removed from the list. For Pacific salmon, the listing unit is the Evolutionarily Significant Unit (ESU), which typically comprises multiple individual populations within a particular geographic region<sup>1</sup>. The Southern Oregon / Northern California Coast (SONCC) Coho Salmon ESU, which represents coho salmon populations found in coastal watersheds from Elk River (Oregon) in the north to Mattole River (California) in the south, was listed as threatened under ESA in 1995 (70 FR 37160).

As part of the recovery planning process, NMFS assembled a group of scientists to serve as a Technical Recovery Team (TRT), whose purpose is to provide a scientific context for identifying necessary actions to help the ESU recover. Among the TRT's responsibilities is the development of biological viability criteria for populations and the ESU that, if met, would indicate when the ESU is recovered and hence form the biological basis for formal delisting criteria<sup>2</sup>.

For the TRT, development of biological viability was a two-step process. The first step was to define the historical population structure within the SONCC ESU. Salmonid species are

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<sup>1</sup> The ESA allows listing not only of species, but also "distinct population segments" of species. The NMFS has developed policies that define distinct population segments as populations or groups of populations that are reproductively isolated from other conspecific population units and that are an important component in the evolutionary legacy of the species. For salmon, NMFS has termed these distinct population segments "Evolutionarily Significant Units" or ESUs (Waples 1991).

<sup>2</sup> Delisting criteria must consider not only the biological status of the listed species, but also the factors that led to the listing of the species in the first place (National Marine Fisheries Service 2006). The biological viability criteria proposed in this document represent what the TRT believes to be the minimum population and ESU characteristics required for the ESU to have a high probability of persisting into the future and at low risk of extinction.

structured hierarchically based on the potential of exchange of individuals between similar components. This can range from sub-populations (e.g., breeding groups), to dependent populations, to independent populations, to population groups, and finally the ESU (Bjorkstedt et al. 2005; Lawson et al. 2006). The likelihood of exchange between components decreases as one builds from sub-populations, to populations, to population groups, to ESUs. Because an ESU is composed of a number of populations with varying features (e.g., habitat size, within population spatial distribution, etc.) and dynamics, an understanding of the biological organization of populations within an ESU and the temporal and spatial scales relevant to this organization is critical to developing meaningful biological viability criteria. A description of biological organization or biological structure of the SONCC ESU is presented in the TRT's report *Historical population structure of coho salmon in the Southern Oregon / Northern California Coasts Evolutionarily Significant Unit* (Williams et al. 2006).

The population structure report (Williams 2006) adopts as its underlying foundation the concept of "independent populations," which McElhany et al. (2000) defined as "any collection of one or more local breeding units whose population dynamics or extinction risk over a 100-year time period are not substantially altered by exchanges of individuals with other populations." This concept is vitally important, as it seeks to delineate population units that should form the appropriate fundamental units of conservation, since a strategy that focuses on units that are too small or those whose persistence depends on immigrants from neighboring populations could lead to a high probability of extinction for the ESU. Williams et al. (2006) describe the historical population structure of coho salmon in the SONCC ESU based on the location and amount of potential coho salmon habitat, with an assumption that the relative

abundance of different populations mirrored the amount of intrinsic habitat potential in each watershed. In general, the SONCC ESU was characterized by 1) small-to-moderate-sized coastal basins in which habitat (as predicted by the intrinsic potential model used in Williams et al. (2006)) was concentrated in the lower portions of the basins, and 2) by three large basins in which some habitat was located in the lower portions of the basins, relatively little habitat was available in the middle portions of the basins, and the greatest amount of habitat was located in the upper sub-basins. This general description of historical coho salmon habitat was then interpreted to define the following historical population structure and distribution:

- 1) Nineteen *functionally independent populations*, defined as those sufficiently large to be historically viable-in isolation and whose demographics and extinction risk were minimally influenced by immigrants from adjacent populations (Tables 1 and 2, Figure 1).
- 2) Twelve *potentially independent populations*, defined as those that were potentially viable-in-isolation, but that were demographically influenced by immigrants from adjacent populations.
- 3) Seventeen small *dependent populations* of coho salmon, which are believed to have had a low likelihood of sustaining themselves over a 100-year time period in isolation and that received sufficient immigration to alter their dynamics and extinction risk.
- 4) Two *ephemeral populations*, defined as populations that were both small enough and isolated enough that they were only intermittently present.

In anticipation of developing viability criteria at the population scale and integrating population information into viability criteria at the ESU scale, Williams et al. (2006) also

identified the diversity and distribution of coho salmon populations that historically existed within the ESU (Figure 2; Plate 1). The TRT organized the independent and dependent populations of coho salmon in the SONCC ESU into diversity strata largely based on the geographical arrangement of these populations and basin-scale environmental and ecological characteristics.

The current report constitutes the second step in the development of biological viability criteria. The report builds on the population structure report (Williams et al. 2006), describing a framework for assessing population and ESU viability for coho salmon in the SONCC ESU. The extinction risk of an ESU depends upon the extinction risk of its constituent populations. Therefore, development of objective and measurable viability criteria for an ESU must start with assessing the viability of its populations. In the Viable Salmonid Populations (VSP) document, McElhany et al. (2000) described four characteristics of populations that should be considered when assessing viability: abundance, productivity, diversity, and spatial structure. The viability of an ESU depends on the appropriate distribution and characteristics of its constituent populations to ensure that longer-term ecological and evolutionary processes are maintained. Consequently, ESU viability criteria should seek to maintain the diversity of an ESU across all or a significant portion of its historical range, to provide redundancy in order to reduce the effects of catastrophic events, and to ensure connectivity among populations across the ESU (Shaffer and Stein 2000). This report proposes criteria for evaluating viability at both the population and ESU levels.

Before presenting a framework to assess the viability of the SONCC coho salmon ESU, we prepare the reader for the fact that no data sets of the appropriate time series or spatial scale are

currently available to assess the viability of any population within the SONCC ESU using our criteria. Therefore, assessing the current viability of the ESU using our approach is likewise impossible. Consequently, our effort has been directed at developing a framework that relates viability to extinction risk. The primary purpose of this framework is twofold: 1) to provide a means by which population and ESU viability can be evaluated in the future, and 2) to propose preliminary biological targets that can assist recovery planners in prioritizing recovery actions.

Our approach is built upon a general approach used for assessing extinction risk developed by the IUCN (Mace and Lande 1991; IUCN 1994) and latter modified by Allendorf et al. (1997) specifically to prioritize conservation actions for Pacific salmonids. We took this path based on the widespread lack of data and recognition that appropriate data sets for performing more sophisticated analyses would not be available for many years. The general criteria thresholds and rules are such that, if met, we believe the ESU would have a high likelihood of being viable. Such an approach is advocated by Shaffer et al. (2002), and was advice given directly to the various TRTs by the Salmon Recovery Science Review Panel (RSRP, 27-29 August 2001 meeting)<sup>3</sup>. As ESU and population-specific research and monitoring occurs, changes in our thresholds could be warranted. This framework will allow recovery planners to have initial population and ESU targets and thresholds of various performance measures that we believe address the VSP concepts of productivity, abundance, spatial structure, and diversity.

Although the appropriate data are not available to assess population viability using the framework developed in this report, the data sets that are available and were used in the most

recent status review (Good et al. 2005) indicated that coho salmon populations in the SONCC ESU continue to be depressed relative to their historical numbers. Good et al. (2005) reported strong indications that breeding groups had been lost from a significant portion of the historical range, and although the 2001 brood year appeared to be strong, it followed a number of relatively weak years. The one exception noted by Good et al. (2005) were the “Rogue River stock” where there were increasing numbers of spawners over the last several years, despite low numbers in 1998 and 1999. The Rogue River stock numbers reviewed by Good et al. (2005) are based on the Huntley Park seine data and represented the whole Rogue Basin, a composite of four independent populations (Lower Rogue River, Illinois River, Middle Rogue and Applegate rivers, and Upper Rogue River populations (Williams et al. 2006)).

We stress that the viability criteria we propose should not be viewed as discrete knife-edge, pass/fail criteria for assessing population extinction risk and ESU viability. Although we have grounded the criteria as firmly as possible in the available science, the specific criteria thresholds are not without scientific uncertainty. Additionally, by their general nature, the criteria do not account for population-specific attributes that might make a population more or less vulnerable to extinction. Hence, there is some potential for the criteria to be overly conservative in some cases, or not conservative enough in others. And finally, any estimation of population parameters will also be subject to some uncertainty. Accordingly, the criteria should be considered as general indicators of viability that are most appropriately viewed with these uncertainties in mind. Furthermore, in evaluating population risk, it is also important to consider each individual

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<sup>3</sup> The Recovery Science Review Panel was convened by the NMFS to help guide the scientific and technical aspects of recovery planning for listed salmon and steelhead species throughout the West Coast. More information about the RSRP and their reports are available at: <http://www.nwfsc.noaa.gov/trt/rsrp.cfm>. The RSRP made specific mention

criterion in the context of other criteria. Populations that fail to satisfy several criteria are likely at greater risk than those that fail to satisfy a single criterion. Likewise, there will be higher uncertainty as to the status of populations that are borderline for several criteria than for those that are marginal for a single criterion.

This framework was developed in part to provide guidance on the types of population and ESU performance measures (i.e., VSP) needed to assess the viability of the SONCC Coho Salmon ESU. Our hope is that through the implementation of rigorously designed and carried out research and monitoring efforts, population-specific viability measures will become available and form the foundation for future population- and SONCC ESU-specific criteria. We feel that this hope is justified when examining situations outside of the SONCC ESU. For instance, the Oregon Coast Coho Salmon ESU has many populations where rigorously designed (e.g., statistically valid) monitoring has been in place for more than 10 years and is providing the TRT and recovery planners for that ESU more population- and ESU-specific data to develop viability criteria. Although differences exist in the final application of the viability criteria proposed for the Oregon Coast and SONCC ESU (i.e., decision support system for Oregon Coast, viability criteria table for SONCC), the approaches and threshold values are quite consistent based on the VSP foundations on which they were developed. In addition, the SONCC and Oregon Coast ESU viability criteria were developed in concert along with the TRT developing recovery criteria for the Central California Coast Coho Salmon ESU, making the viability analyses of these three coho salmon ESUs extremely consistent.

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to the development of relatively simple objective, population-based criteria in their report based on the 27-29 August 2001 meeting.

There was much discussion within the TRT if we should have been more population-specific in terms of developing these viability criteria. In the absence of population- and ESU-specific data, the general conclusion was that such specificity was inappropriate and not approaching viability assessment in a precautionary manner. In addition, with so few monitoring activities of the appropriate nature currently underway in the ESU, we believe our criteria will provide guidance for what types of data should be collected and considered, including direction to the spatial and temporal scales. Through the collection of these data and the use of population viability analyses, it may prove possible for recovery planners to more accurately assess population and ESU viability. In some instances, our criteria may need to be changed, either higher or lower; but in the absence of data from well designed and carried out monitoring activities, the general criteria we propose will serve as a benchmark for recovery planning, as suggested by the RSRP and by Shaffer et al. (2002).

There was some discussion within the TRT for us to be more explicit in terms of the types of monitoring activities required. Given the current state of sampling throughout the ESU and the time required to have time series of the appropriate length, we opted for our more general approach. We believe that the initial efforts to 1) better understand the amount of habitat available to each population, and 2) obtain abundance estimates for each population are going to require a significant increase or redirection of current efforts as well as some trial and error. In terms of implementation (e.g., logistical considerations, methods standardization, site selection, etc.), we will be much better served by having these more general guidelines. Therefore, during the initial phase, more rigid, population-specific requirements may be impractical and overwhelm field efforts, and perhaps discourage needed changes in current monitoring.

However, in some portions of the SONCC ESU where the logistical and institutional infrastructure is in place, implementing more intensive monitoring programs may be possible in the near future.

## 2. Population Viability

### 2.1 Approach

McElhany et al. (2000) define a viable salmonid population as “an independent population... that has a negligible risk of extinction due to threats from demographic variation, local environmental variation, and genetic diversity changes over a 100-year time frame.” Population viability depends on various demographic properties of the population, such as population size, growth rate, variation in growth rate, and carrying capacity (Tuljapurkar and Orzack 1980). McElhany et al. (2000) proposed that four parameters are critical for evaluating population viability: abundance, population growth rate, population spatial structure, and diversity.

Abundance is of obvious importance since, in general, small populations are at greater risk of extinction than large populations, primarily because many processes that affect population dynamics may operate differently in small populations than in large populations (Shaffer 1987; McElhany et al. 2000).

Population growth rate, the productivity over the entire life cycle, and factors that affect population growth rate provide information about how well a population is performing in the various habitats it occupies during the life cycle. Examining population growth rate allows one to assess if populations are able to replace themselves. Populations that consistently fail to replace themselves are at greater risk of extinction than populations that are consistently at or

above replacement levels.

Spatial structure refers to the both the distribution of individuals within a population at a certain life stage throughout the available habitats, as well as the abiotic and biotic processes that give rise to that structure. McElhany et al. (2000) gave two main reasons why spatial structure is important to consider when evaluating population viability: 1) overall extinction risk at longer time scales may be affected in ways not apparent from short-term observations of abundance and productivity, because there can be a time lag between changes in spatial structure and the resulting population-level effects, and 2) spatial population structure affects the ability of a population to respond to changing environmental conditions, and therefore can influence evolutionary processes. Maintaining spatial structure within a population, and the benefits to viability that it imparts, requires appropriate habitat conditions and suitable corridors linking the habitat and the marine environment to be consistently available.

Diversity relates to the variability of phenotypic characteristics such as life histories, individual size, fecundity, run timing, and other attributes exhibited by individuals and populations, as well as the genetic diversity that may underlie this variation. There are many reasons diversity is important in a spatially and temporally varying environment. Three key reasons are 1) diversity allows a species to use a wide array of environments, 2) diversity protects a species against short-term spatial and temporal changes in the environment, and 3) genetic diversity provides the raw material for surviving long-term environmental change (McElhany et al. 2000).

Developing objective, quantitative, and biologically meaningful viability criteria for Pacific salmonid populations is difficult, in part because of a paucity of data and a lack of understanding

of some of the fundamental ecological and biological processes governing populations and ESUs (Wainwright and Waples 1998; Lindley et al. 2007). Although McElhany et al. (2000) provided a useful conceptual framework and guidance for developing viability assessments of Pacific salmon, they did not propose specific quantitative criteria that would allow for assessing the viability of specific populations. Ideally, population-specific criteria could be developed that would take into account differences in the quantity or quality of freshwater habitat, variability in marine survival, or other conditions that influence viability. However, for many populations of Pacific salmonids, including SONCC coho salmon, very little population-specific data are available; therefore, viability criteria are being developed based on biologically relevant criteria that are generic to *Oncorhynchus* species, and in some cases tailored to specific species (this document; (Wainwright et al. In prep.; Spence et al. In prep; Lindley et al. 2007).

Because of the lack of information that would allow for development of population-specific viability criteria, we chose to adopt the general approach of the IUCN (IUCN 1994) as modified for Pacific salmonids by Allendorf et al. (1997). The Allendorf et al. (1997) approach is directed at classifying populations into one of six categories: extinct, extinct in the wild, high risk, moderate risk, and low risk, or data deficient. In Allendorf et al. (1997), evaluation of extinction risk is done by population viability analysis (PVA) or, if data are insufficient to construct a credible PVA, using four surrogate criteria related to population size and trend (see Table 1 in Allendorf et al. 1997).

We adopt the general framework of Allendorf et al. (1997), but propose several modifications and extensions that are important to our particular application. In making these modifications, we draw heavily from similar efforts for California salmonids (Spence et al. In

prep; Lindley et al. 2007).

The most notable change the TRT made to the Allendorf et al. (1997) approach was de-emphasizing the use of PVA for assessing risk. Although PVA can be a valuable tool for assessing extinction risk, there is sufficient uncertainty associated with the use of PVA for evaluating extinction risk that the TRT felt the general criteria (analogous to Allendorf et al.'s "surrogate" criteria) provide the most appropriate indicators of population, especially given the general lack of data appropriate for PVA. The use of PVA is encouraged for developing a better understanding of population-specific processes, and may be used to identify risk not captured by the more general criteria. But unlike Allendorf et al. (1997), we recommend that PVA not be used to supersede the general criteria for low-risk designation.

Like Allendorf et al. (1997), we developed extinction risk categories from high to low defined by various quantitative criteria corresponding to specific extinction risks across specific time horizons (Table 3). We define criteria for a "low risk" category, which was implicit in Allendorf et al. (1997). In addition, we collapse their "very high" and "high" risk categories into a single "high" risk category to simplify analysis. Discriminating between "high risk" and "very high risk" was important to Allendorf et al. (1997) because their emphasis was on prioritizing stocks for conservation. The distinction between "high risk" and "very high risk" was less important for our purposes since either categorization indicates that a population should not be considered viable over short-to-moderate time frames. In practice, the effects of collapsing these two categories are relatively minor, although it does result in a slightly different configuration and implementation of the viability criteria table than that of Allendorf et al. (1997). Specifically, we adopt a rule that the assignment of risk to a population is based on the highest

risk category for any individual risk metric. For example, a population rated at "high risk" based on  $N_e$ , but moderate or low risk for the other metrics would receive the "high risk" rating.

Allendorf et al. (1997) used a similar approach, but added an additional rule where populations that rank at a certain risk level for more than one metric are moved to the next highest risk level when categorizing the population (e.g., a population rated at moderate risk for two metrics is considered at high risk overall). Because of this, the criteria listed in our "high risk" and "moderate risk" categories align themselves with the "very high risk" and "high risk" categories of Allendorf et al. (1997). In practice, a population that satisfies a single criterion (as opposed to two or more) receives the same ranking using either the Allendorf et al. (1997) or our approach. One additional risk category we define is "data deficient" for populations that are believed to still persist, but where data for evaluating risk are partially or entirely lacking. In general, we viewed our configuration of the criteria table to be somewhat simpler to apply to extinction risk and viability.

In addition to these modifications in extinction risk categories, we also add two criteria not found in the Allendorf et al. (1997) framework. First, we add criteria related to the percentage of fish spawning in the wild that are of hatchery origin. These criteria are intended to address potential genetic consequences of hatchery fish on their wild counterparts. Additionally, we add criteria related to spawner density, to address both potential demographic risks associated with very low population densities (i.e., depensation) and concerns about loss of spatial structure and diversity.

In this report, we adopt terms used by the North-Central California Coast Recovery Team for the Central California Coast Coho Salmon ESU (Spence et al. In prep). The term "risk category"

is used to describe the possible status of a population in relation to a specific population characteristic or several population characteristics. Risk categories include extinct, high risk, moderate risk, low risk, and data deficient. A “risk metric” is an attribute of a population that we measure to evaluate risk, and “risk criteria” indicate specific values of a metric that are used to place a population into a risk category. In addition, we use three different terms to describe population size: the number of annual spawners ( $N_a$ ), the number of spawners per generation ( $N_g$ ), and the effective population size ( $N_e$ ). The reason we use express population size estimates as functions of both annual run size and the numbers of spawners per generation reflects the different time scales over which various processes considered by specific criteria occur. For example, many demographic processes operate at an annual time scale whereas it is usually more appropriate to consider genetic processes at generational time scales.

## 2.2 Population viability analysis

Population viability analysis estimates the probability of population extinction or collapse within a given time period (Beissinger and McCullough 2002).. Population viability analysis focuses on how habitat loss, environmental uncertainty, demographic stochasticity, and genetic factors interact to determine extinction risk (National Research Council 1995).

Numerous models for PVA have been proposed (Dennis et al. 1991; Ratner et al. 1997; Lindley 2003), although, the use of PVA has been viewed with skepticism by many scientists (Taylor 1995; Beissinger and Westphal 1998). The practical utility of any viability model depends on the validity of its underlying assumptions (National Research Council 1995). Ellner et al. (2002) discuss the need for some accountability for imprecision in parameter estimates and

its consequences for risk assessments based on PVAs. Reed et al. (2002) reviewed various issues related to the use of PVA as a tool in endangered species management.

Data needed for PVA to estimate extinction risk includes current population abundance, intrinsic population growth rate, habitat capacity, and variation in fecundity, growth, or survival (Belovsky 1987; Lande and Orzack 1988; Lande 1993); see also Beissinger and McCullough (2002) for a review of PVA). Either long-term time series data or intensive population or species-specific data sets (e.g., stage-specific survival rates, etc.) are required for a PVA. In general, short-term studies underestimate extinction risk since they fail to capture the greater temporal variation in population size and demographic parameters used in these models (Reed et al. 2002). We are not aware of data appropriate for PVA for any of the coho salmon populations in the SONCC ESU.

As previously mentioned, there has been concern expressed in the literature over how PVA models are used for making conservation decisions (see Beissinger (2002) for review). The dominant causes of uncertainty in the outcomes predicted by PVA model results include: 1) poor data, 2) difficulties in parameter estimation, 3) weak ability to validate models, and 4) effects of alternative model structures (Beissinger 2002).

There was much discussion within the TRT concerning how best to incorporate PVA into our low-risk criteria in Table 3. One underlying concern was that, given the range of possible approaches to PVA, an analysis might result in an extinction risk <5% within 100 years but may not provide a sufficiently accurate estimate of true extinction risk of a population, especially with the paucity of appropriate data for coho populations in the SONCC ESU. On the other hand, inclusion of the option to use a PVA in addition to the more generic criteria we propose is

intended to encourage a more rigorous population-specific analysis integrating site-specific data. As pointed out in various responses to the Allendorf et al. approach (Wainwright and Waples 1998; Currens et al. 1998) and discussed by others in the literature, PVAs should be just one tool recovery planners use to evaluate extinction risks.

Allendorf et al (1997) proposed the use of population viability analysis (PVA) to estimate extinction risk over a specified time and the use of alternate criteria when an acceptable PVA was not available. For our purposes, we propose the use of our general criteria (analogous to Allendorf et al.'s alternate criteria) to determine if populations are at low risk of extinction (Table 3). We suggest that as data become available, a PVA can be used to gain better understanding of population-specific dynamics and provide insight into possible population-specific characteristics to consider in the context of the general criteria to assess whether a population is at low-risk of extinction. Like Allendorf et al. (1997), PVA can be used to designate a population as at moderate- or high-risk or extinction (Table 3).

Because of the uncertainty expressed by many authors and members of the TRT, we require that all of the general criteria must be met for a population to be declared at low risk of extinction (Table 3). If a credible PVA can be constructed, results should be compared to results of the general criteria we propose, and by comparison of the outcomes, potential limitations of either approach identified and examined. A PVA is not required to determine a low-risk designation, but a PVA alone does not supersede the general criteria. For high-risk and moderate-risk determination, a PVA result alone can be used to establish risk level, although we strongly recommend that the PVA results be compared to results of the general criteria we propose. We also caution against using PVA analysis alone to assess population viability.

Our approach of advocating the use of general criteria rules over a PVA for assessing low-risk status of a population follows closely the recommendations of Shaffer (2002):

“As appealing as PVA models are for doing rigorous risk assessments of species extinction, our collective experience as a community of scientists and managers seems to be that, in most cases, we are probably overdriving our headlights. ... The situation is unlikely to improve quickly. In the meantime, we have only two options for influencing the debate over how much habitat is enough to accomplish conservation: evolution of the status quo, or development of rules of thumb.”

Shaffer (2002) listed the kinds of steps required for the evolution of the status quo to include:

(1) developing standards of data, data analyses, and modeling that constitute an acceptable PVA, (2) model validation through laboratory and field experiments, and (3) long-term research to better understand how populations fluctuate. As part of the recovery planning process, recommendations on research and monitoring needs will be developed by the TRT and will address many of the steps suggested by Shaffer (2002).

The development of rules of thumb suggested by Shaffer (2002) is consistent with the framework the TRT has proposed. To this point Shaffer (2002) states the following:

“But what should be done when there are not data, or when we must prepare for a time horizon that exceeds the ability of the data and models to produce credible forecasts? It seems inescapable that the conservation biology community must, in fact, suggest rules of thumb, or guidelines, that will lead managers to make defensible judgments about how much habitat or what population size is enough to consider a species conserved.”

Shaffer (2002) presented three principles that should be used in developing rules of thumb to make viability judgments without adequate data or models. The three principles are representation, redundancy, and resiliency. Of Shaffer’s three principles, representation and redundancy are most directly captured by our ESU viability criteria whereas resiliency is most directly captured by our spatial structure and diversity criteria at the population level (discussed below). Regardless, the foundation of all of these principles and the fundamental starting point of

our approach is the population unit.

In summary, the TRT strongly recommends that PVA results be compared to results of the other criteria we propose, and that potential limitations of either approach be carefully identified and examined. In addition, any PVA analysis should be subjected to rigorous scientific peer review.

### 2.3 Effective population size / Total population size

These first two extinction risk criteria are intended to address concerns related to loss of genetic diversity within a population. Genetic variability is the source of adaptive potential of a population; loss of genetic variability may affect the ability of a population (or ESU or species) to adapt to environmental change and may reduce survivorship and fecundity (Burgman et al. 1993; Allendorf et al. 1997; Willi et al. 2006). Deleterious genetic effects of small population size are a function of the effective population size ( $N_e$ ), rather than the total number of spawners per generation ( $N_g$ ), or census size.

The effective population size is the number of breeding individuals in an idealized population that would give rise to the same variance in gene frequency under random genetic drift or the same rate of inbreeding as the population under consideration (Wright 1931). The effective population size of a population is usually smaller than the census population size because of variation in individual reproductive success, unequal sex ratios, and temporal variation in population size. (Gall 1987; Burgman et al. 1993; Ardren and Kapuscinski 2003).

Direct estimates of  $N_e$  would be the most relevant measure for evaluating genetic risk to populations, but direct estimates are difficult to obtain from natural populations (Nelson and

Soulé 1987; Waples 2002; Heath et al. 2002). In the absence of a direct estimate of  $N_e$ , an estimate based on the census population size can be made assuming a ratio of effective population size to total population size per generation,  $N_e/N_g$ . Allendorf et al. (1997) assumed a  $N_e/N_g$  ratio of 0.20, citing personal communication with R. Waples (NMFS, Northwest Fisheries Science Center). Later studies with Chinook salmon (Waples 2004) and steelhead (Heath et al. 2002) have reported  $N_e/N_g$  ratios between 0.05 and 0.3. Higher  $N_e/N_g$  ratios have occasionally been reported for salmonids. For example, for a single population of steelhead in Washington, Ardren and Kapuscinski (2003) reported  $N_e/N_g$  ratios over a 18-year period of 0.73 or 0.53, depending on the method of calculation (temporal method versus comprehensive demographic estimate). Nevertheless, we concluded that the 0.20 value proposed by Allendorf et al. (1997) was a reasonably precautionary default value to use for relating total fish abundance to effective population size when no direct estimates are available. The total population size criterion provides an alternative to the effective population size when estimates of effective population size are not available, as will likely be the case with most SONCC coho salmon populations.

We propose three different categories of genetic risk (i.e., high, moderate, low) related to effective population size, which are defined by two thresholds (i.e.,  $N_e = 50$  or  $500$ ). Populations are rated at high risk of extinction at  $N_e \leq 50$  (or  $N_g \leq 250$ ); at this level, populations are believed to be at high risk from random genetic risk such as inbreeding and fixation of deleterious alleles (Nelson and Soulé 1987). Populations with  $50 \leq N_e \leq 500$  (or  $250 \leq N_g \leq 2500$ ) are considered at moderate risk, and populations with  $N_e > 500$  (or  $N_g > 2500$ ) are at low risk of extinction from genetic effects (Table 3). It should be noted, that for SONCC coho salmon with a generation

time of approximately three years, the  $N_g = 250$  would represent an annual abundance ( $N_a$ ) of 83 fish, although effective population size should be considered at the generational time scale.

Effective population sizes less than 50 individuals are believed to be at high risk from random genetic effects such as genetic drift, inbreeding depression, and fixation of deleterious alleles (Frankel and Soulé 1981). There has been an on-going discussion in the literature concerning the use of  $N_e = 500$  as a threshold between low and moderate risk (Allendorf and Ryman 2002). Allendorf et al. (1997) proposed that long-term adaptive potential begins to be compromised at  $N_e = 500$ , although they noted that if populations are reproductively isolated from other populations then the  $N_e$  required to prevent loss of genetic variation might be as much as an order of magnitude greater (i.e.,  $N_e = 5,000$ ; (Nelson and Soulé 1987)). Lande (1995) has suggested that an  $N_e$  of 5,000 rather than 500 may be necessary to maintain normal levels of adaptive genetic variance in quantitative characters under a balance between mutation and genetic drift. This was based on the concern that the models used to derive the  $N_e = 500$  rule assumed all mutations were mildly deleterious, whereas recent work suggests that most mutations with large effects are strongly detrimental, with perhaps only 10% being mildly deleterious. In contrast, the models of Franklin (1980) and Soulé (1980) assume that populations are closed to immigration. Even low levels of immigration (i.e., straying) of as few as one or two individuals per generation, may prevent the loss of alleles through genetic drift (Lacy 1987). For most salmon populations, stray rates among populations at these levels are not unusual, or at least were not under historical conditions. Because violations in the assumptions act in opposition to one another, we accepted the  $N_e = 500$  recommendation of Allendorf et al. (1997)

as a reasonable default criterion for defining the break between populations at low and moderate risk.

Recently, Ardren and Kapuscinski (2003) developed demographic and genetic estimates of  $N_e$  for a steelhead population in Washington and concluded that at  $N_e$  levels between 50 and 500, the population was not losing diversity at a rate fast enough to warrant immediate concerns about inbreeding or loss of heterozygosity. However, by not having an  $N_e$  above 500 the population was unlikely to undergo increases in population genetic parameters, such as additive genetic variation and heterozygosity, that have been positively associated with long-term evolutionary potential. Ardren and Kapuscinski's (2003) results and discussion provide additional support for the use of the 50 and 500 threshold as default values.

The estimate of total population size per generation ( $N_g$ ) we use is based on a harmonic mean of the running sum of adult spawner abundance over the mean generation time. We assume a mean generation time of three years throughout the SONCC ESU. If future research and monitoring indicate population differences in mean generation time, population-specific generation time values should be used.

The total population size per generation is calculated by

$$\tilde{N}_{g(harmonic)} = \frac{1}{\frac{1}{n} \sum_{t=1}^n \frac{1}{N_{g(t)}}}$$

where  $N_{g(t)}$  is the running sum of adult abundance at time  $t$  for a period equal to the mean generation time (three years) and  $n$  is the number of years for which the running sum can be calculated. We recommend a minimum of four generations (i.e., 12 years) since the effects of

these genetic bottlenecks can linger for many generations. We also suggest that, if longer time series of data are available, this criterion be examined to determine if a specific population had been subject to low effective population size at some time in the recent past (e.g., > 12 years). We use the harmonic mean, the reciprocal of the mean of the reciprocals, since it gives greater weight to low values of  $N_g$  and therefore captures our concerns over the potential long-term consequences of a genetic bottleneck on population persistence.

This estimate should be based on counts of naturally spawning fish (including jacks), exclusive of hatchery-origin fish, over a period representing at least four generations (i.e., 12 years). Allendorf et al. (1997) noted that spawner data often exclude jacks; however, jacks may contribute to subsequent generations and therefore need to be accounted for in the total population size estimate, although some adjustment for the relative reproductive success of jacks versus adults may be needed. Recent work by Van Doornik et al. (2002) estimated an effective proportion of 2-year-olds to be 35% in two naturally spawning populations in Washington, suggesting coho salmon should be treated as a species with overlapping generations.

In applying the total population size criteria, there are conditions that may lead to violations in our assumption of  $N_e/N_g = 0.2$  that should be considered. The spatial structure of a population can affect the relationship between census size and effective population size (Whitlock and Barton 1997), as can highly skewed sex ratios, sex-biased differences in dispersal, and substantial among-family variation in survival rates (Gall 1987). Moreover, populations that have undergone a recent bottleneck may have a  $N_e/N_g$  ratio substantially below 0.2, indicating that the population remains at genetic risk even if total population size is large enough to suggest otherwise. Therefore, a population that has experienced a recent bottleneck may require a longer

period of time with relatively high abundance to no longer be considered at risk. Also, it should be noted that Ardren and Kapuscinski (2003) found that a constant  $N_e/N_g$  ratio could not be assumed for wild steelhead population. In the population they examined, Ardren and Kapuscinski (2003) found that  $N_e/N_g$  ratios increased with low numbers of fish, suggesting genetic compensation and increased population productivity during brood years with few spawners can act to reduce demographic and genetic risks of extinction in the population they studied.

## 2.4 Population decline

To address the increased demographic risks resulting from rapid or prolonged declines in abundance resulting in small population size, we have included population decline criteria. The rationale for these criteria is that a severe and prolonged population decline resulting in small numbers of individuals is strong evidence that a population is at risk of extinction. In a variable environment any decline in population capacity proportionally increases the chances of population extinction (Shaffer 1987). We adopt criteria consistent with Allendorf et al. (1997), which include both a downward trend in population size component and a minimum adult run size component.

A population is considered at high risk if it meets any of the following conditions: 1) the population has undergone a decline within the last two generations (i.e., 6 years) to an annual run size ( $N_a$ ) of fewer than 500 spawners, 2) the population has an average annual run size  $N_a > 500$  spawners but is declining at a rate of  $\geq 10\%$  per year over the last two-four generations, or 3) the population currently has an annual average run size of  $N_a > 500$  but is declining at a rate that

would cause  $N_a$  to fall below 500 spawners within two generations (i.e., 6 years). Within this high-risk category, the progeny/parent ratio is less than one, indicating that populations are failing to replace themselves.

If a population has declined to an annual run size below 500 spawners, but the numbers remain stable (i.e., progeny/parent ratio  $> 1$ ) or if  $N_a > 500$  but continues to decline (i.e., progeny/parent ratio  $< 1$ ), though not at a rate that will cause  $N_a$  to fall below 500 spawners within two generations, we consider the population at moderate extinction risk. Populations with annual run sizes  $N_a > 500$  adults and no apparent decline over the last two generations are considered at low extinction risk for this criterion. We note that these run sizes are distinct from  $N_e$  or census population size based on a  $N_e/N_g$  ratio used in evaluating effective population size; effective population size estimates represent abundance over an entire generation. For these population decline criteria we are evaluating the annual abundance ( $N_a$ ). For consistency with Allendorf et al. (1997), we chose to use the value of 500.

The abundance threshold we adopt from Allendorf et al. (1997) is in the absence of information on intrinsic growth rate (i.e., growth rate when populations are released from competition at low population density). Modeling efforts to estimate extinction probability are often very sensitive to assumptions about intrinsic growth rate and environmental stochasticity (Goodman 1987; Lande 1993). For instance, a population with a high intrinsic growth rate and a low variance in the growth rate might have a relatively low extinction risk at  $N_a < 500$ , but a high probability of extinction if intrinsic growth rates were low and exhibited large variation. Recovery planners should be cautious relaxing the thresholds for this criterion, especially when  $N_a < 500$ . Rigorous monitoring and evaluation is needed to support changes this threshold.

Recently Lindley (2003) suggested that a minimum of 30 years of data are likely required to obtain unbiased estimates of variance in population growth rate within reasonable confidence limits.

The population decline criteria require the calculation of two parameters, the mean annual spawner abundance ( $N_a$ ) and the population trend ( $T$ ). We recommend using the geometric mean of the most recent four generations (i.e., 12 years) to estimate annual population abundance. The geometric mean is slightly more conservative than the arithmetic mean. This estimate should be based on naturally spawning fish, exclusive of hatchery-origin fish. This estimator is consistent with those used in recently published status reviews (Good et al. 2005).

Population trend should be evaluated for all populations to ascertain whether mean abundance is declining at a rate  $> 10\%$  year, or if continued decline at the current rate would result in the annual spanwer abundance to drop below the 500 fish threshold. In addition, examination of population trend well assist in the evaluation of populations where  $N_a < 500$  and the population appears to be stable. The population trend is estimated by the slope of the regression of the generational running sum of abundance,  $N_{g(t)}$  (ln-transformed), over two generations of generational running sums of abundance (Table 4, Figure 3). The population decline criteria are intended to capture recent, relatively rapid declines in abundance. Over longer periods of time, populations that decline at less than 10% annually may still be at high risk of extinction. In the SONCC ESU, there are few existing time series of population abundance spanning longer than 10 years. In these cases, long-term trends should be evaluated independently of the proposed population decline criteria.

Examination of the trend will provide context when considering population declines, especially for populations that are below an  $N_a$  of 500, but appear to be stable in their abundance. Interpretations of population trends can be greatly influenced by the tendency of salmon populations to naturally fluctuate at time scales ranging from annual to decadal or longer, which can lead to highly variable estimates of trend. Since few time series of the appropriate length currently exist for coho populations in the SONCC ESU, estimates of  $T$  will likely be based on relatively short time series of  $N_a$ . Interpretation of  $T$  needs to be made in the context of marine and freshwater survival during time period being examined. If this population trend is negative (i.e., a negative slope), the population is failing to replace itself and population abundance should not be considered stable.

## 2.5 Catastrophe

Catastrophes are large environmental disturbances that produce rapid and dramatic declines in population abundance (Shaffer 1987; Lande 1993). These types of disturbances are different than smaller stochastic environmental events that result from the continuous disturbances that affect population growth rate (e.g., ocean conditions, interannual climate variability). Catastrophes can occur across a range of spatial scales, from localized disturbances affecting a few miles of stream and therefore only a portion of a population, to those that may encompass the habitats of several populations (e.g., large wildfires). For the purpose of developing population viability criteria, we focus on catastrophes that affect all or a substantial portion of a population, producing a rapid and dramatic decline in population abundance. (The issue of

larger-scale catastrophes is treated in ESU viability criteria.) In general, populations are at increased risk of extinction following a major reduction in abundance.

The criteria of Allendorf et al. (1997) defined a very high-risk situation as a 90% decline in population abundance within one generation and a high-risk situation as “any lesser but significant reduction in abundance resulting from a single event or disturbance.” Their purpose was to capture situations where a population had experienced a drastic shift from a low risk to a higher risk level. Although Allendorf et al. (1997) built upon the IUCN criteria (Mace and Lande 1991), they differed from those of the IUCN, which proposed that average population declines over two to four generations of 50%, 20%, and 10% corresponded to critical, endangered, and vulnerable status, respectively. Allendorf et al. (1997) did not elaborate on why they differed from the IUCN criteria, but they did acknowledge that Pacific salmon and trout often exhibit substantial natural variation in abundance. Based on Allendorf et al.’s (1997) general statements concerning variability in abundance, we concluded that they considered that the declines of the magnitude specified in the IUCN criteria would fall within the range of natural variation observed in salmonids and therefore adopted their more strict criteria.

We adopt the criteria as presented by Allendorf et al. (1997). Populations that have experienced a 90% decline in abundance within one generation are considered to be at “high risk” of extinction. Populations that have experienced a lesser but significant decline are considered at moderate risk. Although Allendorf et al. (1997) do not specifically define what constitutes a “lesser but significant,” we consider events such as a loss or near loss of a year class resulting from a catastrophic disturbance to be an example of a lesser but significant decline that would warrant classifying a population as at moderate risk of extinction. Such losses

may be particularly important to coho salmon, which have a relatively fixed three-year life cycle with minimal overlap in generations. Events such as landslides, fire, severe flood or drought, chemical spills, or some other catastrophic event affect populations differently and the risk associated with these different events can vary substantially depending on the specific circumstances such as year class strength. Because of this, we do not propose specific numeric thresholds for moderate risk and instead recommend that risk be evaluated on a case-by-case basis.

There may be instances where a population exhibits a clear precipitous decline in abundance or suffers a major loss or alteration of habitat (e.g., chemical spill affecting an entire year class, migration blockage, or some other catastrophic event) that should result in an immediate elevated risk designation, even in the absence of a longer time series of adult spawner abundance data. In addition, there may be situations where a longer time series indicates that a population has experienced a catastrophic decline in abundance at some time in the past. In such cases, and consideration should be given to the response of the population following the catastrophic decline (see Figure 4). For example, certain type of catastrophic disturbances (e.g., wildfires, landslides that block access to habitats) may have long-term effects on carrying capacity, which may preclude a population from recovering rapidly. Other catastrophic disturbances (e.g., chemical spills), while they may strongly influence a year class or two, may not involve fundamental long-term shifts in habitat capacity, allowing for more rapid recovery as conditions improve.

The estimator we propose for catastrophic decline ( $C$ ), is the maximum proportional change in abundance from one generation to the next:

$$\hat{C} = \text{maximum} \left( 1 - \frac{N_{g(t)}}{N_{g(t-2h)}} \right)$$

where  $N_{g(t)}$  is the running generational sum of adult spawners in year  $t$  (i.e.,  $N_{a(t-2)} + N_{a(t-1)} + N_{a(t)}$ ), and  $N_{g(t-2h)}$  is the running sum at time  $t-2h$ , where  $h$  is the mean generation time (i.e., three years for SONCC coho salmon).

The estimation of this criterion is based on the maximum proportional change in abundance from one generation to the next, although this is not simply based on the previous generation. Instead, we propose that the proportional change be based on the generation sum from two generations previously and therefore require a time series of adult spawner abundance (naturally spawning fish, exclusive of hatchery fish) of at least three generations (i.e., 9 years). For example, since coho salmon have a mean generation time of three years, the value calculated would be the sum of adult abundance for years 7, 8, and 9 (the most recent) divided by the sum of abundance for years 1, 2, and 3. We base these criteria on a time series over three generations because the value is highly influenced by the pattern of abundance during the transition period of high abundance to a period of low abundance as a result of it being based on a running sum of abundance. For example, consider the two time series of abundance presented in Figure 5. The “open-square line” depicts a situation where a population averaging around 50,000 spawners in years 1 through 13, drops in a single year to an average of 5,000 spawners in years 14 through 30. The “solid-square line” illustrates a similar scenario, but the decline occurs over a generation (3 years), rather than in a single year. If the running sum in the most recent generation were used in the denominator of the estimator, the value of  $C$  would exceed the 90% decline threshold only for the scenario where the decline occurred over one year (open-square line). In the second

scenario, the intermediate abundances in years 14 and 15 moderate the value of C, such that the 90% criterion is never exceeded, despite the order of magnitude drop in abundance that occurred over a single generation (3 years). Our estimator would capture both scenarios as a catastrophic decline, regardless of whether the decline occurred over a single year or a full generation.

## 2.6 Spatial Structure and Diversity

As noted earlier, the spatial structure and diversity of populations can contribute to population persistence. Allendorf et al. (1997) included criteria for effective population size, which addresses to some degree potential loss of diversity associated with small population size; however, they provided no criteria that deal with potential loss of spatial structure or the loss of diversity that may result when populations no longer inhabit the range of environments that were historically occupied. Consequently, the TRT proposes as an addition to the Allendorf et al. framework criteria for spawner density that are intended to address these two population attributes (i.e., spatial structure and diversity).

Spatial structure and diversity influence population viability by spreading risk, both spatially and temporally, in addition to contributing to the resiliency of populations to various disturbances. The spatial arrangement of suitable spawning and rearing habitat within a watershed can be dynamic through time as a result of periodic disturbances that create a mosaic of varying habitat conditions (Reeves et al. 1995). Coho salmon distributed throughout a diversity of habitat conditions within a basin may be able to persist through periods of localized disturbance, with fish from areas not impacted by the disturbance available for recolonization (Parvinen et al. 2003; Kun and Scheuring 2006). A diversity of habitat conditions throughout a

basin also results in exposure to a range of environmental conditions, which can lead to expression of greater phenotypic and genotypic diversity (Williams and Reeves 2003). Over the short term, greater phenotypic diversity (e.g., variation in adult or juvenile migration timing, length of freshwater residence, etc.) can help spread ecologic risk (den Boer 1968). Over longer temporal scales, genetic diversity provides the material that a population draws upon to adapt to changes to the environment. As a population departs from its historical patterns of distribution and abundance as a result of habitat loss or degradation, the probability of persistence likely decreases, though numerous factors will determine how far a population can depart from historical conditions and still remain viable (Willi et al. 2006).

In contrast, populations that have been severely reduced in number may be subject to directional demographic processes that result in increased extinction risk. At very low densities, populations can experience a reduction in per capita growth rate with declining abundance, a phenomenon referred to as *depensation*. Depensation occurs when populations are reduced to very low densities and per capita growth rates decrease as a result of a variety of mechanisms (e.g., failure to find mates and therefore reduced probability of fertilization, failure to saturate predator populations, etc.; (Liermann and Hilborn 2001). Depensation results in a negative feedback that accelerates a decline toward extinction.

The spatial structure and diversity criteria, based on spawner densities, vary with the size of the watershed the population occupies. Other viability metrics we have proposed are fixed values (i.e., effective population size and population decline criteria). The spatial structure and diversity criteria accounts for historical differences in the total habitat available, the relative capacity of the habitat, the role of spatial structure and diversity in population persistence, the role of

nutrient subsidies and physical actions of spawning in maintenance of ecosystem productivity, or the possibility of depensation when few individuals within a population are sparsely distributed across the available habitat. For example, an effective population size of 500 (or  $N_g$  of 2500) may be appropriate for a small basin and also represent an appropriate density of spawners, but the same number of fish spread throughout a large basin may be at a high risk of extinction. The spatial structure and diversity criteria would require the larger basin to have more spawners than the smaller basin, although both basins would still have the 500-fish threshold for effective population size.

At this point it might be useful to step back and consider how one might view population viability, and perhaps provide a useful context for considering spawner density and its use to capture our concerns with spatial structure and diversity. Within the framework of viable salmon populations, population viability can be considered from two distinct but equally important perspectives (Spence et al. In prep). The first perspective involves defining the minimum viable population size (MVP) for which a population can be expected with some specified probability to persist over a specified period of time (Soulé 1987; Ralls et al. 2002). From this perspective, the minimum viable population size can be thought of as identifying the approximate lower bounds for a population where risks associated with demographic stochasticity, environmental stochasticity, severe inbreeding, and long-term genetic losses are negligible (Soulé 1987). This view of viability asks where a population is likely going in the future, but not necessarily where it has been in the past; in respect to genetic diversity, criteria related to a fixed MVP threshold size are intended to guard against further erosion of genetic diversity, but not necessarily consider diversity that may have been lost (Spence et al. In prep).

The second perspective in which viability can be considered is in respect to how a population is currently functioning in relation to its historical viability (Spence et al. In prep). From this perspective, the historical patterns of abundance, productivity, spatial structure, and diversity form the reference conditions about which there is high confidence that the population had a low extinction risk. This perspective takes a longer term and more broad view and asks how a population functioned in its historical context (e.g., what roles did spatial structure and diversity play in population persistence?) and what role the population played in relation to other populations within an ESU (Spence et al. In prep). As a population departs from historical conditions, its extinction risk likely increases and its functional role with respect to ESU viability diminishes.

Both perspectives are captured by our criteria. Our use of spawner density addresses the longer-term risks associated with loss of spatial structure and diversity that are important both for population resilience and therefore persistence, and the ability of populations to fulfill their roles within the ESU and contribute to ESU viability.

In developing spawner density criteria to capture issues related to spatial structure and diversity, we made the following four assumptions. First, the historical distribution and abundance of spawners for an independent population represent reference conditions where extinction risk was likely low. Populations most likely tended towards a general carrying capacity, and the spatial structure, diversity, and productivity of the ecosystem resulted in low extinction risk in the absence of large-scale catastrophes. Second, the further a population diverges from historical conditions, the greater the extinction risk and the greater the uncertainty of the population's viability. The more restricted and fragmented the distribution of individuals

within a population, the higher the extinction risk. Some departure from historical conditions (e.g., diminished habitat conditions, reduced spatial distribution, loss of access to portions of habitat) may have little influence on population persistence, but the more these conditions diverge from historical conditions the greater the uncertainty of the population's viability. Third, the size of the population and its historical distribution largely determines how far it can deviate from historical conditions and remain viable. The thresholds we propose based on a minimum amount of potential habitat capacity (IP-km) required for viability-in-isolation are based on the assumption that, under historical conditions, populations were at or near carrying capacity. For example, a comparable percentage reduction of habitat would be less likely to increase extinction risk in a large watershed than it would in a small watershed where the attendant reduction in abundance and distribution would be more likely to move the population below levels required for viability. This is especially true for small populations that are near the IP-km threshold for independence (Williams et al. 2006). Fourth, at extremely low densities populations are at a greater risk of extinction resulting from depensation.

This potential habitat capacity, intrinsic potential (IP), is based on results from a Geographic Information Systems (GIS) model to predict the IP of coho salmon habitat (Burnett et al. 2003; Burnett et al. 2007). Its application to SONCC coho salmon was described by Williams et al. (2006). In brief, the model predicts the potential for a stream reach to exhibit habitat characteristics suitable for a specific life-history stage as a function of the underlying geomorphic and hydrologic characteristics of the landscape. A stream reach is a section of stream or river approximately 50 to 200 m in length, and is generally defined with respect to geomorphological features. Mean gradient, mean annual discharge, and valley constraint of

stream reaches are used in the analysis. These characteristics are selected on the basis of being effectively constant features of the landscape that directly control the processes that create, alter, and maintain essential features of salmon habitat.

Specifically, IP is calculated as the geometric mean of suitability scores, which range from 0-1 and describe the potential that a stream reach with a specific value for a given characteristic will exhibit suitable habitat. These scores are generated by mapping the values for each of the three habitat characteristics onto suitability curves. The IP model itself has the structure of a limiting factors analysis, in that a low suitability score for a single habitat characteristic can greatly reduce (or eliminate) the potential for suitable habitat. We used this approach to generate predictions of IP for habitat of coho salmon using approaches developed by Burnett et al. (Burnett et al. 2003; Burnett et al. 2007). The estimates of stream km based on IP model represent the total length of contiguous stream reaches with an IP score  $> 1$ . The IP score for each reach is weighted by the reach length (IP score  $\times$  reach length), and the values for all reaches accessible by a given population are summed to obtain an estimate of IP-km for that population. (see pages 12-14 in Williams et al. (2006) for more details of the IP model and how the historical distribution was derived).

The first three assumptions listed above relate directly to how far a population can diverge from historical conditions and remain viable, and therefore, provide our low risk-threshold. In practice, determining the low-risk threshold is also a difficult task since the scientific basis for quantitatively relating spatial structure, diversity, and ecosystem productivity to extinction risk is currently limited. The last assumption directly relates to the establishment of the high-risk threshold where we are concerned with the density at which depensation is likely to occur in

coho salmon populations. Detecting depensation in salmonid populations is likewise difficult (Liermann and Hilborn 1997; Liermann and Hilborn 2001). Despite these uncertainties, we believe reasonable criteria can be developed from published literature and our general principles.

As discussed above, the spatial structure and diversity criteria based on spawner density define two thresholds. The first addresses concerns resulting from depensation at very low densities and distinguishes between populations at high versus moderate risk. The second addresses concerns with spatial structure, diversity, and productivity and distinguishes between populations at moderate versus low risk. As previously discussed, defining a density at which depensation is likely to occur is extremely difficult since it can be highly variable and because of the limited number of spawner-recruit datasets where observations have been made at low abundances (Liermann and Hilborn 1997). Despite these difficulties, there have been numerous efforts to define thresholds at which depensation appears to occur in salmonids, and many of these efforts have focused on coho salmon (Chilcote 1999; Barrowman et al. 2003). In general, these and other studies have found little evidence of depensation in coho salmon unless densities were less than 1 female/km. Assuming a 50:50 sex ratio, this equates to 2 adult/km. In Chilcote (1999), it is suggested that coho salmon populations in the lower Columbia River were unlikely to recover if densities fell below 2.4 adults/km. The Oregon Coast Workgroup of the ONCC (Wainwright et al. In prep.) concluded that at spawner densities of 0.61 spawners/km (1 spawner/mile) demographic risks were very likely to be significant.

Based on these studies, we set the threshold for high risk of depensation as those populations with an average spawner density of fewer than 1 adult per IP-km. We chose to use IP-km in the denominator in order to account for potential differences in habitat quality among watersheds.

This was based on an assumption that IP-km provides a reasonable measure of the relative productive potential of a watershed. For basins with similar IP-km but different total km, the average density based on adults/km might be expected to be lower in the less productive watershed, perhaps leading to a greater depensation risk. However, we assume that in most cases fish will be distributed somewhat according to habitat quality. In general, the ratio of IP-km:total km averages about 0.60 for watersheds within the SONCC ESU. Consequently, the OR Coast Workgroup value of 0.6 spawner/km translates to approximately 1 spawner/IP-km, the criterion we use.

The value for our low-risk threshold varies as a function of population-specific estimates of habitat capacity (Figure 6). For the smallest watersheds capable of supporting an independent population in the SONCC coho salmon ESU, low-risk populations are those exceeding 40 spawners/IP-km. For larger watersheds, the density requirement decreases with increasing capacity (i.e., IP-km) to a minimum of 20 spawners/IP-km based on our assumption that larger populations can diverge farther from historical conditions before extinction risk is substantially increased.

The low-risk criteria are based on the assumption that populations historically occurred, on average, at something close to the natural carrying capacity of the system they occupied. Based on their viability-in-isolation analysis, Williams et al. (2006) defined the minimum threshold of potential habitat (expressed as IP-km) required for a population to be considered viable-in-isolation to be 34 IP-km for SONCC coho salmon. Therefore, for populations in the smallest watersheds (in terms of IP-km) capable of supporting viable independent populations to remain viable, the population must function at something close to its historical carrying capacity. Any

reduction in capacity (either loss of access to portions of the historical habitat, or reduction in the productive capacity of the historical habitat) would drop the population below the threshold for viability (i.e., independence). Based on this reasoning, the average spawner density at historical carrying capacity serves as a reasonable basis for establishing the low-risk threshold for the smallest watersheds.

The TRT relied heavily on the work of Bradford et al. (Bradford et al. 2000) to address the difficult task of estimating the threshold for the low-risk criteria based on spawner density. Bradford et al. (2000) fit a model to 14 historical data sets of coho salmon from the Pacific Northwest and found that, on average, a density of 19 females/km is required to fully seed freshwater habitats with juveniles. Assuming a sex ratio that is slightly biased for males, we rounded the number to approximately 40 spawners/km for watersheds with a minimum IP required to be considered independent.

The establishment of the low risk threshold of 40 spawners/IP-km for the smallest populations was largely dictated by the threshold for viability-in-isolation proposed by Williams et al. (2006) and supported by empirical data and various modeling efforts reported in the literature. To accommodate our assumption that for larger populations, a comparable percentage reduction in habitat is less likely to result in a substantial increase in extinction risk as it would in smaller populations, we assume that a population with ten-fold additional habitat potential than the smallest population requires an average spawner density of half that of the smallest population. This captures our general conclusion that the larger the historical population, the more it can depart from historical conditions and remain viable. The function we propose to capture this is a linear decline in required density between 40 spawners/IP-km in the smallest

populations to 20 spawners/IP-km in the watersheds with greater than 10-fold the habitat potential of the minimum watershed (i.e., IP-km > 340, Figure 6).

The development of this latter reference point was by the NCCC TRT (Spence et al. In prep.) after much review and discussion, and although it is based largely on expert opinion, it provides results that are qualitatively consistent with the general hypotheses relating watershed size and density to spatial structure, diversity, and other factors that influence population persistence. The benefits of our approach for these criteria are that it establishes a population-specific abundance that is scaled to the amount of potential habitat and avoids the use of fixed abundance criteria. In addition, this approach captures the elements of spatial structure and diversity that contribute to viability without rigidly defining what the spatial structure must look like. For instance, in a large watershed the density criteria could be satisfied either by having fish distributed throughout the watershed at moderate densities or by having high densities in approximately half the habitat. Each of these scenarios has advantages and disadvantages from a population persistence perspective. For example, moderate densities spread throughout a watershed may be more resilient to localized disturbances than populations with more localized groups of fish at densities near carrying capacity densities. Conversely, localized areas of high productivity may be critical for population persistence during periods of unfavorable environmental conditions (Nickelson and Lawson 1998). Currently, we lack the appropriate data to make more spatially explicit criteria on spatial structure, but believe our approach captures the essence of the spatial structure and diversity elements outline by McElhany et al. (2000) for viable salmon populations. Future research and monitoring may allow for the development of explicit population-specific distribution criteria.

For the high-risk threshold that captures our concerns related to depensation, we propose an estimate of average spawner density (spawners/IP-km) in the three consecutive years of lowest abundance (i.e., a moving three year average) within the last four generations (i.e., 12 years). Mathematically, we express this as follows:

$$\hat{D}_{dep} = \frac{\min\left(\frac{N_{g(t)}}{3}\right)}{X}$$

where  $N_g$  is the running generational sum of annual spawner abundance at time  $t$  and  $X$  is the estimate of potential habitat capacity in IP-km for the watershed containing the population. We propose averaging the spawner density over three years within the last four generations versus a single year or over all years in order to have an indicator that is sensitive to a population that is at risk of depensation, without being too sensitive to natural fluctuations in abundance. For example, a population that experiences a single year of low abundance may be at minimal risk of falling into an accelerating pattern of depensation, especially if there is some overlap of generations, which may be able to rebound more rapidly after a poor year. Averaging over all years might lead to a few relatively good years masking a general pattern of very low spawner abundance. The use of the lowest three consecutive years looks for recurring evidence of population numbers sufficiently low that there is heightened potential for depensation. A concern we have, but not captured in this metric, is the possibility of having two relatively healthy brood cycles of a coho population masking the third brood cycle that is facing a high depensation risk. Taking the running sum may mask such a scenario, although such a situation may be captured by our population decline criteria or the “lesser but significant decline” element of the catastrophe

criteria. For now, we recommend that recovery planners examine such situations on a case-by-case basis.

For the low-risk threshold, we propose the arithmetic mean of adult spawner density for all years over the last four generations, expressed as adult spawners/IP-km:

$$\hat{D}_{ssd} = \frac{1}{4h} \sum_{t=1}^{4h} \frac{N_a}{X}$$

where  $N_a$  is annual spawner abundance and  $X$  is the estimate of potential habitat capacity in IP-km for the watershed containing the population, and  $h$  is the mean generation time for the population. For coho salmon, we assume a mean generation time of three years throughout the SONCC ESU, in the absence of population-specific estimates of generation time. The estimated density is then evaluated against thresholds that are a function of population-specific estimates of potential habitat capacity or IP-km (Table 5, Figure 6).

Estimates of density can be obtained from two different sampling approaches. First, a direct weir count, or a count at some other fish passage facility, divided by the number of stream IP-km accessible in the watershed. The count can be either a total fish count if all upstream migrating fish are counted or a total population estimate, if only a portion of upstream migrating fish are captured and the proportion can be accurately estimated. Both of these types of counts estimate annual run size, although they must represent the whole population being considered. Second, randomized spawner surveys can provide a total population estimate, which can then be divided by the total accessible IP-km to obtain an average density over the entire watershed.

## 2.7 Hatchery influence

Hatchery programs and the presence of cultured fish can impose various biological problems on salmon populations that include genetic and evolutionary risks, demographic risks, ecological risks, and problems due to the behavior, health status, or physiology of hatchery fish (National Research Council 1996). Specific genetic risks can include four fundamentally different adverse consequences: extinction, loss of within-population variability, loss of among-populations variability, and domestication (Busack and Currens 1995). Demographic risks can include direct risk, such as when wild adults are captured for hatchery broodstock, or indirect, such as when releases of large numbers of hatchery fish lead to excessive harvest on wild fish in mixed-stock fisheries. Ecological risks can include competition between hatchery and wild fish, predation on wild fish by hatchery fish or by predators attracted to abundant hatchery fish, and transmission of diseases between hatchery and wild fish.

The evaluation of the various types of potential impacts of hatchery fish on wild fish is extremely difficult, as many of these impacts can be highly context-dependent. For example, the potential for negative competitive interactions depends on numerous factors, including the number and size of hatchery fish released, the size of the recipient population relative to carrying capacity, and the timing and location of release, among other factors. Likewise, genetic impacts of hatchery fish on wild fish depends on the origin of hatchery broodstock, broodstock collection practices, mating and rearing protocols, the duration of exposure to hatchery practice, the number of hatchery fish spawning in the wild, and a host of other factors. Consequently, an analysis of various impacts, in many instances, is best done on a case-by-case basis when specifics of past, present, and future hatchery operations can be considered. For the SONCC

Coho Salmon ESU, we are not aware of any rigorous studies currently available that demonstrate no or negligible ecological or genetic effects resulting from current or past hatchery operations.

Despite these difficulties, the TRT felt that criteria addressing the potential genetic risks of hatchery fish were warranted, as there exists a substantial literature on these risks (Chilcote 2003; Eium and Fleming 2001; Goodman 2005; National Research Council 1996). For our purposes, we consider a population to be at least at moderate risk if the fraction of naturally spawning fish that are of hatchery origin, as determined by appropriately designed surveys, exceeds 5%. Populations are at low risk if no or negligible ecological or genetic effects resulting from current or past hatchery operations can be demonstrated.

The TRT recommends a general low-risk threshold of 5% with recognition that although the appropriate low-risk threshold value is difficult to determine, there is very strong support in the literature for a precautionary approach when considering impacts of hatchery fish on wild populations (National Research Council 1996) and attempts to mitigate negative effects of habitat degradation by releasing hatchery-produced fish (Eium and Fleming 2001). Several researchers have suggested that even minimal contribution of hatchery fish can pose a risk to wild populations. In their review of literature data, Eium and Fleming (2001) reported that numerous studies have found interactions between wild and released salmonids resulting from current hatchery practices may be detrimental to the recipient wild populations. Chilcote (2003) suggested that if his findings for steelhead held for other salmonids, an effective method to increase the productivity of natural populations and associated conservation benefits may be to minimize the frequency of hatchery fish in natural spawning populations. Goodman (2005) found through modeling of phenotypic evolution of integrated hatchery and wild spawning

programs a potential for substantial erosion of natural spawning fitness. Moreover, Goodman (2005) cautions that the modeling shows that the depression of natural spawning fitness increases with the magnitude of the hatchery contribution, it would be good to determine empirically whether specific policy caps on the amount of hatchery contribution can limit the fitness erosion to a tolerable level. In their “Native Fish Conservation Plan” (Oregon Department of Fish and Wildlife 2003), the Oregon Department of Fish and Wildlife proposes that at least 90% of the spawners within a population must be naturally produced and not hatchery produced fish (with some exceptions).

We do not propose specific metrics for assessing genetic risk beyond the general low-risk threshold of 5%, acknowledging the uncertainty in quantitatively relating the risk of hatchery fish to extinction risk by a single ESU-wide threshold might not be appropriate. For instance, various best management procedures may be in place that might reduce the risk and allow for a more liberal threshold level. Also, an appropriately planned and executed phase-out of a hatchery program might also allow for a more liberal threshold level if the targeted endpoint is a viable wild spawning population, recognizing that in some situations hatchery programs may play a role in population recovery. The 5% threshold we propose is a default value for recovery planners in the absence of any population-specific research and monitoring that would more directly measure risks, both genetic and the host of others previously mentioned potential impacts of hatchery fish on the wild population of interest. In addition, this 5% is made in the absence of information concerning the hatchery practices in place. Factors that contribute to genetic risk such as origin of broodstock (e.g., within basin, within ESU, etc.), broodstock collection procedures, and mating and rearing protocols should be considered by recovery

planners in assessing the impacts of hatchery fish. In addition, hatchery operations change over time. The legacy of past hatchery practices as well as current hatchery protocols should be considered on a case-by-case basis.

The types of analyses used to assess the impacts of hatchery contributions on a population should include a suite of considerations. Although not exhaustive, the list below provides an example of the types of questions that must be addressed before a credible determination of hatchery risk can be made:

- Is the productivity of the naturally spawning component of a population consistent with demographic viability?
- Are there changes in characteristics of the integrated population that indicate reduced ability of naturally spawning component to persist in the absence of hatchery production?
- Is the hatchery prone to outbreaks of disease that would place a wild or integrated hatchery-wild population at heightened demographic risk?
- If wild fish are being used for broodstock, is the remaining wild population at heightened risk?

We also suggest an approach that builds upon the efforts by the Interior Columbia Basin TRT (Interior Columbia Basin Technical Recovery Team 2005a; Interior Columbia Basin Technical Recovery Team 2005b), which recognized that the risk associated with hatcheries is heavily influenced by not only fraction of hatchery fish spawning in the wild, but also the degree of genetic similarity between hatchery and wild broodstock, and the number of generations over which the impact has occurred.

## 2.8 Population Viability of SONCC coho salmon populations

As discussed in the introduction of this document, there are currently almost no data at the appropriate spatial scale and or temporal scale (i.e., enough years of data from present back 9 to 12 years) to assess the viability of coho salmon populations in the SONCC ESU. This is not to say there are not efforts currently underway within the SONCC ESU to collect various types of data on coho salmon. While these programs may be important for answering specific questions about the ecology of coho salmon populations in general, or local patterns of trend and abundance, they are generally not sufficient for assessing status and trends at the population level. A benefit of many of the projects currently underway is that they will provide recovery planners some insight into various biological, ecological, and logistical considerations needed to develop a more comprehensive monitoring program for coho populations in the SONCC ESU.

For recovery planners, our criteria (Tables 3) provide guidance on the types of data of interest, the spatial scale needed (population unit), and the temporal length of time series required to assess viability, even with the very general approach we have proposed. The approach we have taken provides a framework that can assist recovery planners develop recovery strategies and evaluate progress toward recovery, providing targets that can help prioritize recovery efforts within the SONCC ESU (Table 5). We also provide a summary sheet that, if appropriate data were available, would provide the requisite information to assess ESU viability using our approach (Table 6). Clearly, considerable data are needed to perform a relatively simple and objective assessment of status. Recovery planners need to be extremely cautious concerning efforts to assemble various incomplete or incompatible data sets in hopes of obtaining the various population-specific values we have proposed to assess viability.

Conclusions from such efforts can be based on incomplete data or sampling efforts that are not representative of the greater population of interest and could lead to erroneous conclusions about current viability.

Past status reviews of coho salmon in the SONCC ESU have necessarily relied on existing data that were not intended to provide population-level estimates of abundance. In the most recent federal status review, Good et al. (2005) found that coho salmon populations in the SONCC ESU continue to be depressed relative to their historical number and that there were strong indications that breeding groups had been lost from a significant portion of the historical range. Although the appropriate data are lacking to assess population viability using the framework we have proposed, data available at the present and used by Good et al. (2005) are in agreement with an earlier assessment (Weitkamp et al. 1995) that SONCC coho salmon are likely to become endangered in the foreseeable future. In these status reviews, uncertainty that arises from concerns about the rigor of sampling designs, representation of the sampling unit, and length of time series are recognized and factors into final conclusions about status. The framework we have proposed provides a consistent and robust means for evaluating status that reduces these uncertainties, in addition to providing the population and ESU targets needed for recovery planning.

### 3. ESU Viability

The viability of an ESU depends on several factors including the number and status of populations, spatial distribution of populations, the characteristics of large-scale catastrophic risk, and the collective diversity of the populations and their habitat (Lindley et al. 2007). In anticipation of developing viability criteria at the population scale and integration of population information into viability criteria at the ESU scale, groups of populations spanning the diversity and distribution that historically existed within the ESU were identified by the TRT (Williams et al. 2006); Plate 1). These groups or “diversity strata” reflect diversity of (potential) selective environments. The TRT organized the independent and dependent populations of coho salmon in the SONCC ESU into diversity strata largely based on the geographical arrangement of the populations coupled with multivariate analysis of basin-scale environmental and ecological characteristics (Williams et al. 2006).

#### 3.1 Characteristics of a viable ESU

The foundation of ESU viability is built upon the ability of populations to function in an integrated manner and persist across the landscape. This integration includes dispersal among populations (i.e., connectivity) and a diversity and distribution of habitat types and conditions that allow for the expression of a range of life-history types (Williams and Reeves 2003). For an ESU to be viable, the number and distribution of its constituent populations would exist in a balance between connectivity through dispersal and isolation from common catastrophic risks;

viable populations needs to be in close enough proximity to ensure connectivity, but not so close as to have a high likelihood of being affected by the same catastrophic event.

For an ESU to persist, populations within the ESU must be able to track changes in environmental conditions. When the location or distribution of a species' (or ESU's) habitat changes, a species can avoid extinction either by adapting genetically to the new environmental conditions or by spatially tracking the environmental conditions to which it is adapted (Pease et al. 1989). A species or ESU persists in places where it is able to track environmental changes, and becomes extinct if it fails to keep up with the shifting distribution of suitable habitat (Thomas 1994). These changes in environmental conditions can range spatially and temporally. Spatial scales can range from localized impacts (e.g., affecting one or a few populations) to regional impacts from catastrophic events (e.g., drought) affecting all populations within a diversity stratum. Temporal scales can range from a site specific impact resulting from a short-term, albeit catastrophic, event (e.g., landslide, temporarily blocking passage on a large mainstem river), to interannual variability of various environmental conditions (e.g., upwelling conditions, annual precipitation patterns), to long-term environmental changes such as climate change that have the potential to impact all populations within the ESU.

Interaction among populations of an ESU buffers against catastrophic loss of many populations, maintains long-term demographic and evolutionary processes through connectivity, and maintains sufficient diversity so that the ESU has the evolutionary potential to deal with changing environmental conditions. Some populations need to have sufficiently large numbers of individuals to disperse and provide the needed connectivity among populations, including strays to dependent populations, thereby increasing connectivity throughout the ESU. Thomas (1994)

proposed that population declines in the larger habitats may be as worrying as local extinction in small patches. Recently, Isaak et al. (2007) found that for Chinook salmon in central Idaho, the size and connectivity of habitat were the strongest predictors of occupancy. In addition, these large populations increase the overall abundance of the ESU and also provide an additional buffer against catastrophic disturbance. For these purposes, functionally independent and potentially independent populations are essential to ensure connectivity based on their historical functional roles. Dependent populations that occupy smaller watersheds also contribute to connectivity and provide an essential contribution to ESU viability.

A critical element identified in conservation planning is representation (Groves et al. 2002; Carroll et al. 2006). By representation, we mean establishing populations across the full range of historical potential habitats and ecological settings within the ESU, to the greatest extent possible. By incorporating representation into recovery criteria, it is recognized that a single population may not represent species recovery, even if it is large enough to be significantly resilient to extinction (Carroll et al. 2006).

The TRT worked from the assumption that the historical conditions are the conditions where we are most certain that the ESU could have persisted for long periods of time (e.g., hundreds of years). Therefore, the historical population structure of coho salmon populations that make-up the SONCC ESU provides a template against which the outcome of our proposed ESU viability assessment could be evaluated. As with population viability, our certainty concerning ESU viability decreases as conditions depart from historical conditions. The historical population structure of SONCC coho salmon proposed by Williams et al. (2006) recognized different functional roles that populations played within the historical ESU (i.e., functionally independent,

potentially independent, dependent, and ephemeral) and a general structure for ESU viability represented by the diversity strata (Plate 1). The TRT's approach to developing their picture of historical population structure was built upon the foundation that persistence of the ESU was contingent upon the ability of its populations to track changes in the environment. As discussed previously for population viability, there are certain attributes that populations must exhibit to also track and persist with changes in the environment, but these are mostly at smaller spatial scales and temporal scales than those considered at the ESU level.

### 3.2 ESU Criteria

In our proposed scenario for a viable ESU, we do not list specific sets of populations that must be viable to have a viable ESU. Instead, we provide a set of rules that will result in certain configurations of populations that we believe will result in a viable ESU (Table 7). The rules we propose are intended to capture our objectives of maintaining diversity throughout the ESU, providing connectivity among populations to maintain long-term demographic and genetic processes, and providing a buffer against potential catastrophic risks. Our overarching goal in developing these rules is that we desire an appropriate number and arrangement of populations that allows for the populations to track changes in environmental conditions and therefore be viable. Shaffer et al. (2002) and (Shaffer and Stein 2000) discuss three principles that should be considered when making viability judgments: representation, redundancy, and resiliency. We have incorporated these principles into our viability framework at the ESU level and believe they are equally pertinent even when adequate data or models are available. Though the availability of more complicated models might reduce our uncertainty at the population viability level, and

we believe capturing many of the issues related to representation, redundancy, and resiliency is best done at the ESU spatial and temporal scale.

### 3.2.1. *Representation*

All diversity strata must be represented for the ESU to be considered viable. As previously discussed, the diversity strata were largely based on the geographical arrangement of the populations and basin-scale environmental and ecological characteristics. A represented diversity strata is one that is viable (as defined below). By requiring all diversity strata to be represented for the ESU to be viable, the range of environmental conditions historically available have a greater chance of being included and thereby a substantial portion of the historical diversity of the ESU. In addition, given the geographic make-up of the diversity strata, by requiring all strata be viable helps ensure that the ESU persists throughout a significant portion of its historical range. And finally, because of the linear arrangement of populations along portions of the SONCC ESU, representing each stratum also ensures that connectivity across the entire ESU is maintained.

### 3.2.2 *Redundancy and Connectivity*

Three additional viability criteria are propose that seek to provide a hedge against catastrophic risk by ensuring redundancy of viable populations, and to ensure connectivity within diversity strata and throughout the ESU. In developing recovery plans, the configuration of populations should emphasize historical populations that were the foundation of the ESU.

- a. At least two (2) or fifty percent (whichever number is greater) of historical functionally independent and potentially independent populations in the stratum must be demonstrated to be at low risk for extinction according to the population viability criteria developed in this report.*

**- and -**

- b. Total abundance within the populations selected to satisfy this criterion must meet or exceed 50% of that historically predicted for the diversity stratum based on the Spatial Structure and Diversity criteria (spawner density) for population viability.*

For a diversity stratum to be viable, the above criteria requires that at least two or 50% of the independent populations (functional independent or potentially independent) within a diversity stratum, whichever number is greater, must be viable and that the abundance of these viable populations collectively must meet or exceed 50% of the abundance predicted for the stratum by the Spatial Structure and Diversity (i.e., the total abundance predicted for all independent populations within the stratum when at low risk). Any functionally independent or potentially independent populations that contribute to the aggregate stratum abundance must be viable.

By requiring at least two populations or 50% of populations within a strata be viable, redundancy is provided to help guard against loss of significant diversity as a result of catastrophic events. Requiring selected populations meet or exceed 50% of the abundance predicted for the stratum accomplishes two important objectives (Table 8). First, ensures that sufficient numbers of fish are present throughout the stratum and ensures connectivity. Second, meeting the stratum abundance requirement means that proposed recovery scenarios must

include historically independent populations that, by virtue of their size and location, were disproportionately important to ESU function and persistence. Historically large populations were critical not only because their large size and spatial distribution imparted greater resiliency in the face of longer-term environmental change, but also because they were major sources of dispersers, which likely affected the dynamics of adjacent populations (Table 8). In practical application, recovery of the larger independent populations will satisfy both portions of this criterion. In some strata, particularly those with several historically independent populations that were smaller in size, the second portion of this criterion requires recovery planners to focus efforts on restoring some presumably historically larger independent populations (even if only portions of the populations habitat are still accessible or suitable) or restore additional smaller independent populations to a sufficient degree for the within stratum abundance to satisfy the second part of this criterion. In other words, a stratum cannot be considered recovered by focusing exclusively on the smallest historical populations within it.

*c. All remaining populations, including dependent populations and independent populations (functional and potentially independent populations) that are not expected to meet the low-risk threshold, must exhibit occupancy patterns that indicate sufficient immigration is occurring from the “core populations”.*

This criterion addresses our concerns that connectivity be maintained or reestablished among populations, and we propose that recovery planners place a high priority on populations that are remnants of historically independent populations with a minimum standard that most historically

independent populations should be at no greater than moderate risk of extinction (i.e., not at high risk) when evaluated as independent populations. This requirement, that these populations be at moderate risk, requires a higher standard for occupancy than just presence of individuals. It should be recognized that these independent populations no longer fulfill their historical role within the ESU, but they can play a critical role in connectivity and have the potential for representing critical components of the evolutionary legacy of the ESU.

Dependent populations also fulfill a role in maintaining connectivity among populations, especially in situations where historically independent populations are at high risk or have been extirpated. Dependent populations contribute to maintaining genetic diversity within a stratum and provide strays (i.e., colonizers) that can reduce the genetic and demographic risks to adjacent independent populations and support restoration of these independent populations. After reestablishment of adjacent independent populations, dependent populations can provide some resilience, that is, buffer these larger populations against future disturbances. Dependent populations can also play the role of “steppingstones,” allowing individuals and thereby populations to track changes in environmental conditions.

*d. The distribution of extant populations, both dependent and independent, need to maintain connectivity across the stratum as well as with adjacent strata.*

As with the previous criterion, this criterion also reflects our concern with maintaining, or reestablishing, connectivity among populations within and among diversity strata. In order to ensure that connectivity is maintained within and among diversity strata, there may be specific

populations (and the watersheds that they occupy) that are essential for filling what might otherwise be substantial spatial gaps within a diversity stratum. These watersheds may contain populations considered to have been historically dependent on immigration; therefore, ensuring such populations persist requires that their source populations be also at sufficient status to maintain connectivity.

This distribution criterion addresses the need for a rather continuous set of populations across the ESU to allow for connectivity and provide a buffer against catastrophes and smaller disturbance events. We were not able to develop a specific distance measurement between sites that would fulfill this criterion. However, it would be preferable to avoid having all the viable independent populations and occupied dependent populations isolated into a single geographic region of a stratum. Typically, the further the distance among populations, the less likely immigration occurs. Genetic data supports this strong concordance of geographic and genetic distances, often referred to as isolation by distance (Bucklin et al. 2007). While data are too limited to provide a specific geographic distance at this time to help guide recovery planners, we propose this more general criterion of avoiding a clumped distribution of viable populations across the ESU. There are currently few data on stray rates and stray distances for Pacific salmonids along the southern Oregon and California coastal region to provide strict guidance on how close adjacent populations should be to maintain connectivity. Studies from other regions of Chinook salmon (Hard and Heard 1999), pink salmon (Wertheimer et al. 2004), chum salmon (Tallman and Healey 1994), and Atlantic salmon (Jonsson et al. 2003) suggest that the majority of straying occurs within a few tens of kilometers from their natal stream (or stream of release). Assuming coho salmon in the SONCC ESU exhibit similar tendencies, unoccupied gaps along

the coastline of more than 20 to 30 km may be sufficient to limit patterns of dispersal and connectivity.

### 3.3 Viability of the SONCC Coho Salmon ESU

We were not able to assess the viability of the SONCC Coho Salmon ESU with the quantitative approach proposed in this report, due to data limitations. There are almost no data with which to assess the status of any of the populations of coho salmon in the SONCC ESU described by Williams et al. (2006). Good et al. (2005) concluded that SONCC coho salmon were likely to become endangered in the foreseeable future, this conclusion being consistent with an earlier assessment (Weitkamp et al. 1995). Although there are little data, the information that is available for SONCC coho salmon indicates the component populations are in decline and strongly suggests the ESU is at risk (Weitkamp et al. 1995; California Department of Fish and Game 2002; Good et al. 2005).

## 4. Summary and Recommendations

The purpose of this report was to develop a framework for evaluating the viability of coho populations and the greater SONCC Coho Salmon ESU. This framework is built upon a foundation proposed by Allendorf et al. (1997) of relatively simple criteria and rules that have modest data requirements. The framework proposed in this report parallels efforts for other ESUs in California (Spence et al. In prep.; Lindley et al. 2007). In addition, the approach is consistent in its underlying conceptual approach (e.g., VSP) and the general level of criteria

thresholds used for assessing viability of Oregon Coast Coho Salmon ESU (Wainwright et al. In prep.). The framework we propose consists of criteria and rules that are based in part on expert opinion and judgment, although we have attempted to provide support for our efforts from the general literature, specific references for Pacific salmonids, and when available, references specifically for coho salmon. Although the approaches used by this and other TRTs are to some degree based on expert judgment and subject to considerable uncertainty, the conclusions are not particularly sensitive to the exact threshold values of the criteria (Lindley et al. 2007).

As previously discussed, there are insufficient data to assess the risk of any coho population within the SONCC ESU, and therefore, we cannot assess the viability of the ESU using the quantitative approach developed in this report. As recovery planning proceeds, ambitious research and monitoring programs will need to be initiated, as will long-term restoration solutions. The TRT is preparing a summary of research and monitoring needs for SONCC coho salmon. In the meantime, Lindley et al. (2007) identified some very important actions that should be done as soon as possible for Central Valley salmonids, several of which are pertinent to SONCC coho salmon: (and listed in no particular order):

1. Secure all extant populations. Although the SONCC ESU is far short of being viable, extant populations, even if not currently viable, may be needed for recovery.
2. Begin collecting distribution and abundance data throughout the SONCC ESU. These data are fundamental for developing effective recovery actions and eventual delisting.
3. Minimize straying from hatcheries to natural spawning areas. The goal of maximizing diversity within populations and the ESU is eroded with even low levels of straying from hatchery populations to wild populations.

4. Begin conducting critical research on climate change and its potential impact to SONCC coho salmon. In their report on endangered and threatened fishes in the Klamath River basin, the National Research Council (2004) discussed the need for resource managers to develop strategies to cope with climate change in the Klamath Basin, suggesting an adaptive management approach given the uncertainty in the magnitude and timing of climate change. They go on to state that climate change could make temperature an even greater issue than it is currently for the future of salmonids in the Klamath basin, and suggest that interior basins like the Shasta River with headwater and groundwater recharge areas at high elevation may be more resilient than most stream reaches in the event of increases in temperature and changes in precipitation patterns (National Research Council 2004). It is not unreasonable to suspect that similar scenarios could occur throughout the SONCC ESU, particularly in the Rogue and Eel river basins that penetrate far inland and to high elevations.

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Table 1. Independent and dependent populations of coho salmon in the Southern Oregon/Northern California Coast Coho Salmon ESU, including integrated intrinsic potential (IP) values with 21.5 °C air temperature mask (locations where IP was reduced by temperature mask have pre-mask values in parentheses) from Williams et al. (2006). Population types include functionally independent (FI), potentially independent (PI), dependent (D), and ephemeral (E) populations. Basins with integrated IP < 1.2 km with temperature mask were excluded from analyses. Basins in italics contained an integrated IP TM value < 5 km and are not included by name or number in subsequent analyses. ID numbers are for reference to other tables and figures in this document and Williams et al. (2006).

Basin	ID	IP (km)	Population type		
			FI	PI	D
Elk River	1	62.64	X		
Mill Creek	2	7.25			X
Hubbard Creek	3	17.94			E <sup>a</sup>
Brush Creek	4	5.68			X
Mussel Creek	5	6.06			X
Euchre Creek	6	32.31			E <sup>a</sup>
<i>Greggs Creek</i>		3.40			X
Rogue River		2344.58 (2547.01)			
Lower Rogue River	7a	80.88		X	
Illinois River	7b	589.69	X		
Middle Rogue and Applegate rivers	7c	758.58 (760.67)	X		
Upper Rogue River	7d	915.43 (1115.77)	X		
Hunter Creek	8	14.63			X
<i>Myers Creek</i>		3.45			X
Pistol River	9	30.23			X
<i>Sand Creek</i>		1.62			X
<i>Thomas Creek</i>		1.36			X
Chetco River	10	135.19	X		
Winchuck River	11	56.50		X	
<i>Gilbert Creek</i>		1.80			X
Smith River	12	385.71	X		
Elk Creek	13	17.38			X
Wilson Creek	14	18.80			X
<i>False Klamath Cove</i>		2.17			X
Klamath-Trinity		2247.74 (3048.37)			
Lower Klamath River	15a	204.69	X		
Middle Klamath River	15b	113.49 (178.59)		X	
Upper Klamath River	15c	424.71	X		
Salmon River	15d	114.80 (145.90)		X	
Scott River	15e	440.87	X		
Shasta River	15f	531.01 (606.86)	X		
South Fork Trinity River	15g	241.83 (342.47)	X		

Basin	ID	IP (km)	Population type		
			FI	PI	D
Lower Trinity River	15h	112.01 (170.49)		X	
Upper Trinity River	15i	64.33 (533.79)	X		
<i>Fern Canyon</i>		3.66			X
<i>Squashan Creek</i>		2.66			X
<i>Gold Bluff</i>		2.88			X
Redwood Creek	16	151.02	X		
McDonald Creek	17	5.44			X
Maple Creek/Big Lagoon	18	41.30		X	
Little River	19	34.20		X	
Strawberry Creek	20	5.71			X
Norton/Widow White Creek	21	8.54			X
Mad River	22	152.87	X		
Humboldt Bay tributaries	23	190.91	X		
Eel River - Full		1459.81 (1773.37)			
Lower Eel and Van Duzen rivers	24a	393.52	X		
South Fork Eel River	24b	481.11	X		
Mainstem Eel River	24c	143.90 (156.73)		X	
North Fork Eel River	24d	53.97 (83.54)		X	
Middle Fork Eel River	24e	77.70 (252.77)		X	
Middle Mainstem Eel River	24f	255.50 (281.31)	X		
Upper Mainstem Eel River	24g	54.11 (124.39)		X	
<i>Fleener Creek</i>		3.87			X
Guthrie Creek	25	14.16			X
<i>Oil Creek</i>		3.09			X
Bear River	26	47.84		X	
<i>Singley Creek</i>		3.40			X
<i>Davis Creek</i>		1.71			X
<i>Domingo Creek</i>		1.36			X
McNutt Gulch	27	5.90			X
Mattole River	28	249.79	X		

<sup>a</sup> – Hubbard and Euchre creeks were designated as Ephemeral populations.

Table 2. Population unit boundaries of sub-basins in the Rogue, Klamath, and Eel rivers for SONCC Coho Salmon ESU.

Basin	ID	Population Unit	Boundaries
Rogue	7a	Lower Rogue River	Mouth of Rogue upstream to confluence of Illinois River.
	7b	Illinois River	
	7c	Middle Rogue River	Confluence of Illinois River upstream to confluence of Evans Creek (non-inclusive); includes Applegate River.
	7d	Upper Rogue River	Evans Creek (inclusive) upstream to IP limit.
Klamath	15a	Lower Klamath River	Mouth of Klamath upstream to confluence with Trinity River.
	15b	Middle Klamath River	Confluence of Trinity River upstream to Portuguese Creek (inclusive in Middle Klamath); Seiad and Grider creeks in Upper Klamath basin.
	15c	Upper Klamath River	Portuguese Creek (non-inclusive) upstream to Spencer Creek (inclusive).
	15d	Salmon River	
	15e	Scott River	
	15f	Shasta River	
	15g	S. Fk. Trinity River	Confluence of Trinity River is lower boundary.
	15h	Lower Trinity River	Confluence of Klamath River upstream to confluence with North Fork Trinity River (non-inclusive).
	15i	Upper Trinity River	Confluence of North Fork Trinity River (inclusive) upstream to Ramshorn Creek (inclusive).
Eel	24a	Lower Eel/Van Duzen River	Mouth of Eel River upstream to confluence with South Fork Eel River.
	24b	South Fork Eel River	
	24c	Mainstem Eel River	Confluence of South Fork Eel River upstream to confluence with Middle Fork Eel River.
	24d	North Fork Eel River	
	24e	Middle Fork Eel River	

Basin	ID	Population Unit	Boundaries
	24f	Middle Mainstem Eel River	Confluence of Middle Fork Eel River upstream to Tomki Creek (inclusive), upstream in Outlet Creek and tributaries to IP limit.
	24g	Upper Mainstem Eel River	Eel River upstream of confluence of Tomki Creek (non-inclusive) to IP limit.

Table 3. Viability criteria for assessing extinction risk for populations of coho salmon in the Southern Oregon/Northern California Coast ESU. For a given population, the highest risk score for any category determines the populations overall extinction risk. Modified from Allendorf et al. (1997).

Criterion	Extinction risk		
	High	Moderate	Low
	- any One of -	- any One of -	- all of -
Effective population size	$N_e \leq 50$	$50 < N_e \leq 500$	$N_e > 500$
	- or -	- or -	- or -
Population size per generation	$N_g \leq 250$	$250 < N_g \leq 2500$	$N_g \geq 2500$
Population decline	Precipitous decline <sup>a</sup>	Chronic decline or depression <sup>b</sup>	No decline apparent or probable
Catastrophe, rate and effect	Order of magnitude decline within one generation	Smaller but significant decline <sup>c</sup>	Not apparent
Spatial structure and diversity (spawner density per IP km)	$N_a < 1$	$1 < N_a < \text{MRSD}^d$	$N_a > \text{MRSD}^d$
Hatchery influence			Hatchery fraction < 5%
<i>- in addition to above -</i>			
Extinction risk from PVA	> 20% within 20 years	> 5% within 100 years	< 5% within 100 years <sup>e</sup>

<sup>a</sup> - Decline within the last two generations to  $N_a \leq 500$  spawners OR  $N_a > 500$  but decline of 50% over last 2 generations **and** the current rate of decline would result in an  $N_a < 500$  in the next 2 generations.

Historically small but stable populations not included.

<sup>b</sup> - Annual spawner abundance  $N_a$  has declined to  $N_a \leq 500$  spawners, but now stable OR number of adult spawners ( $N_a$ ) > 500 but declining at a rate of < 50 % over last 2 generations.

<sup>c</sup> - Annual spawner abundance decline in one generation < 90% but biologically significant (e.g., loss of year class).

<sup>d</sup> - MRSD = minimum required spawner density is dependent of the amount of potential habitat available.

Figure 6 summarizes the relationship between spawner density and IP-km.

<sup>e</sup> – For a population to be considered at low-risk of extinction, all of the criteria must be satisfied. In addition, a PVA can be also included for consideration, but must estimate an extinction risk  $< 5\%$  within 100 years *and* all the other criteria must be met. If discrepancies exist between PVA results and other criteria, results need to be thoroughly examined and that potential limitations of either approach be carefully identified and examined.

Table 4. Example data set of annual adult spawner counts ( $N_a$ ) and calculation of generational abundance (3-year running sum) and natural log of generational running sum used to calculate population trend. The  $\ln(\text{running sum})$  is based on the " $N_a + 1$ " in the event there is  $N_a = 0$ . In this example, annual abundance counts fell below the 500 level five times within the past 2 generations and there was a 32% decline over that period (2001 = 1648; 2006 = 1118); note that the generational sum is used to calculate the percent change over two generations and data over eight years (1999 – 2006) are required.

Year	$N_a$	$N_a + 1$	Running sum (3-year)	$\ln(\text{running sum})$
1993				
1994				
1995	810	811		
1996	950	951		
1997	690	691	2453	7.805
1998	820	821	2463	7.809
1999	700	701	2213	7.702
2000	525	526	2048	7.625
2001	420	421	1648	7.407
2002	600	601	1548	7.345
2003	475	476	1498	7.312
2004	390	391	1468	7.292
2005	400	401	1268	7.145
2006	325	326	1118	7.019

Table 5. Specific viability criteria of coho salmon populations in the SONCC ESU. Percent lost IP-km represents the amount of habitat currently located upstream of dams. Depensation threshold represents the minimum number of spawners required to avoid depensation risk; spawner threshold is the minimum number of spawners required for a population to be considered at low-risk for the spatial structure and diversity threshold (based on spawner density value, fish/IP-km, multiplied by IP-km).

Population unit	IP-km			Depensation	Spawner density	Spawner threshold
	historical	current	% Lost	Threshold (fish)	(fish/IP-km)	low risk
Elk River (1)	62.64	59.22	5	63	38	2400
Lower Rogue River (7a)	80.88	80.77	< 1	81	37	3000
Illinois River (7b)	589.69	547.05	7	590	20	11800
Mid. Rogue/Applegate rivers (7c)	758.58	603.90	20	759	20	15200
Upper Rogue River (7d)	915.43	508.21	45	915	20	18300
Chetco River (10)	135.19			135	33	4500
Winchuck River (11)	56.50			57	39	2200
Smith River (12)	385.71			386	20	7700
Lower Klamath River (15a)	204.69			205	29	5900
Middle Klamath River (15b)	113.49			113	34	3900
Upper Klamath River (15c)	424.71	240.45	43	425	20	8500
Salmon River (15d)	114.80			115	35	4000
Scott River (15e)	440.87			441	20	8800
Shasta River (15f)	531.01	426.88	20	531	20	10600
South Fork Trinity River (15g)	241.83			242	26	6400
Lower Trinity River (15h)	112.01			112	35	3900
Upper Trinity River (15i)	64.33	41.17	36	64	37	2400
Redwood Creek (16)	151.02			151	32	4900
Maple Creek/Big Lagoon (18)	41.30			41	39	1600
Little River (19)	34.20			34	41	1400
Mad River (22)	152.87			153	32	4900

Population unit	IP-km			Depensation	Spawner density	Spawner threshold
	historical	current	% Lost	Threshold (fish)	(fish/IP-km)	low risk
Humboldt Bay tributaries (23)	190.91			191	30	5700
Low. Eel/Van Duzen rivers (24a)	393.52			394	20	7900
South Fork Eel River (24b)	481.11			481	20	9600
Mainstem Eel River (24c)	143.90			144	33	4700
North Fork Eel River (24d)	53.97			54	39	2100
Mid. Fork Eel River (24e)	77.70			78	37	2900
Mid. Mainstem Eel River (24f)	255.50	242.93	5	256	25	6500
Upper Mainstem Eel River (24g)	54.11	0.53	99	54	39	2100
Bear River (26)	47.84			48	40	1900
Mattole River (28)	249.79			250	26	6500

Table 6. Population viability evaluation worksheet for SONCC Coho Salmon independent populations. ( $N_e$ : effective population size;  $N_g$ : number of spawners per generation; T: population trend, slope of the regression of the generational running sum of abundance; C: catastrophic decline, the maximum proportional change in abundance from one generation to the next;  $D_{dep}$ : depensation risk threshold, average spawner density in lowest years;  $D_{ssd}$ : spatial structure and diversity, adult spawner density). Where criteria are assigned “na”, data were insufficient in quality or lacking (dd = data deficient).

Population unit	PVA result	Effective population size		Population decline		Catastrophe	Density		Hatchery H	Risk category
		$N_e$	$N$	S	Rate		$D_{dep}$	$D_{ssd}$		
Elk River (1)										dd
Lower Rogue River (7a)										dd
Illinois River (7b)										dd
Mid. Rogue/Applegate rivers (7c)										dd
Upper Rogue River (7d)										dd
Chetco River (10)										dd
Winchuck River (11)										dd
Smith River (12)										dd
Lower Klamath River (15a)										dd
Middle Klamath River (15b)										dd
Upper Klamath River (15c)										dd
Salmon River (15d)										dd
Scott River (15e)										dd
Shasta River (15f)										dd
South Fork Trinity River (15g)										dd

Population unit	PVA	Effective population size		Population decline		Catastrophe	Density		Hatchery	Risk
	result	$N_e$	$N$	S	Rate		$D_{dep}$	$D_{ssd}$	H	category
Lower Trinity River (15h)										dd
Upper Trinity River (15i)										dd
Redwood Creek (16)										dd
Maple Creek/Big Lagoon (18)										dd
Little River (19)										dd
Mad River (22)										dd
Humboldt Bay tributaries (23)										dd
Low. Eel/Van Duzen rivers (24a)										dd
South Fork Eel River (24b)										dd
Mainstem Eel River (24c)										dd
North Fork Eel River (24d)										dd
Mid. Fork Eel River (24e)										dd
Mid. Mainstem Eel River (24f)										dd
Upper Mainstem Eel River (24g)										dd
Bear River (26)										dd
Mattole River (28)										dd

Table 7. Summary of ESU viability criteria for SONCC coho salmon.

ESU viability characteristic	Criteria
Representation	All diversity strata must be viable
Redundancy and Connectivity	
a.	The greater of two (2) OR 50% of the independent populations within a stratum must be viable AND
b.	Total abundance within the populations selected to satisfy the 2 or 50% rule must meet or exceed 50% of the total spawner abundance predicted for the stratum based on the Spatial Structure and Diversity criteria
c.	All dependent and independent populations not expected to meet low-risk threshold within a stratum must exhibit occupancy indicating sufficient immigration is occurring from the “core populations”.
d.	The distribution of extant populations, both dependent and independent, need to maintain connectivity across the stratum as well as with adjacent strata.

Table 8. Diversity strata of the SONCC Coho Salmon ESU including population type (F: functionally independent, P: potentially independent, D: dependent, and E: ephemeral), population unit name (ID number), the low-risk Spatial Structure and Diversity threshold value of number of spawners for independent populations (both functionally and potentially independent populations), and the number of spawners needed to satisfy the 50% of the total number of spawners in a strata needed to meet stratum viability (see text for details).

Stratum	Pop. type	Population unit	Low risk N <sub>a</sub>
Northern Coastal Basins	F	Elk River (1)	2,400
	P	Lower Rogue River (7a)	3,000
	F	Chetco River (10)	4,500
	P	Winchuck River (11)	2,200
	D	Mill Creek	
	E	Hubbard Creek	
	E	Euchre Creek	
	D	Hunter Creek	
	D	Pistol River	
50% total stratum N <sub>a</sub>			6,050
Central Coastal Basins	F	Smith River (12)	7,700
	F	Lower Klamath River (15a)	5,900
	F	Redwood Creek (16)	4,900
	P	Maple Creek/Big Lagoon (18)	1,600
	P	Little River (19)	1,400
	F	Mad River (22)	4,900
	D	Elk Creek	
	D	Wilson Creek	
	D	McDonald Creek	
	D	Strawberry Creek	
	D	Norton/Widow White Creek	
50% total stratum N <sub>a</sub>			13,200
Southern Coastal Basin	F	Humboldt Bay tributaries (23)	5,700
	F	Low. Eel/Van Duzen rivers (24a)	7,900
	P	Bear River (26)	1,900
	F	Mattole River (28)	6,500

Stratum	Pop. type	Population unit	Low risk N <sub>a</sub>
	D	Guthrie Creek	
	D	McNutt Gulch	
		50% total stratum N <sub>a</sub>	11,000
Interior – Rogue River	F	Illinois River (7b)	11,800
	F	Mid. Rogue/Applegate rivers (7c)	15,200
	F	Upper Rogue River (7d)	18,300
		50% total stratum N <sub>a</sub>	22,650
Interior - Klamath	P	Middle Klamath River (15b)	3,900
	F	Upper Klamath River (15c)	8,500
	P	Salmon River (15d)	4,000
	F	Scott River (15e)	8,800
	F	Shasta River (15f)	10,600
		50% total stratum N <sub>a</sub>	17,900
Interior - Trinity	F	South Fork Trinity River (15g)	6,400
	P	Lower Trinity River (15h)	3,900
	F	Upper Trinity River (15i)	2,400
		50% total stratum N <sub>a</sub>	6,350
Interior - Eel	F	South Fork Eel River (24b)	9,600
	P	Mainstem Eel River (24c)	4,700
	P	North Fork Eel River (24d)	2,100
	P	Mid. Fork Eel River (24e)	2,900
	F	Mid. Mainstem Eel River (24f)	6,500
	P	Upper Mainstem Eel River (24g)	2,100
		50% total stratum N <sub>a</sub>	13,950

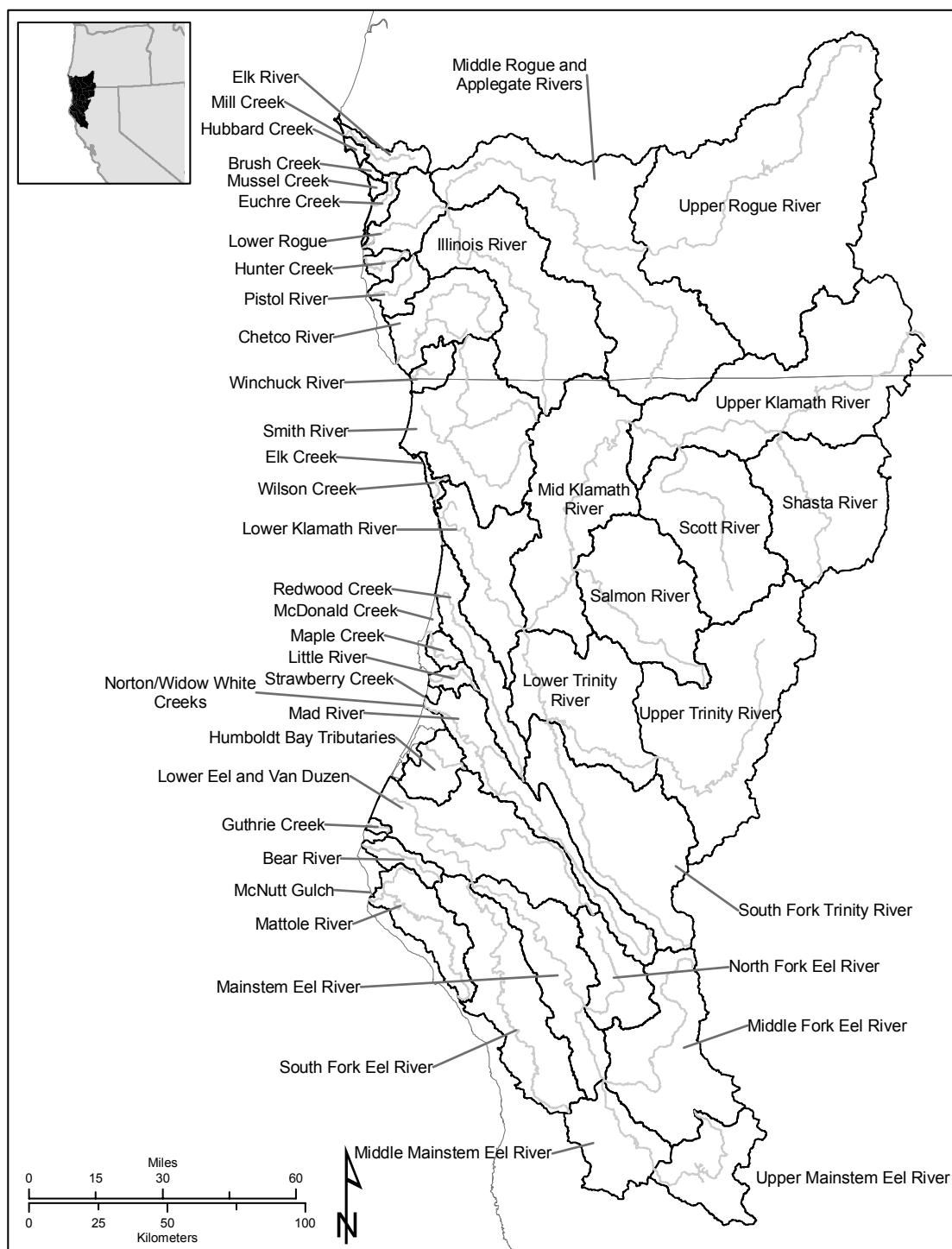


Figure 1. Southern Oregon/Northern California Coho Salmon ESU and independent populations proposed by the TRT (Williams et al. 2006)

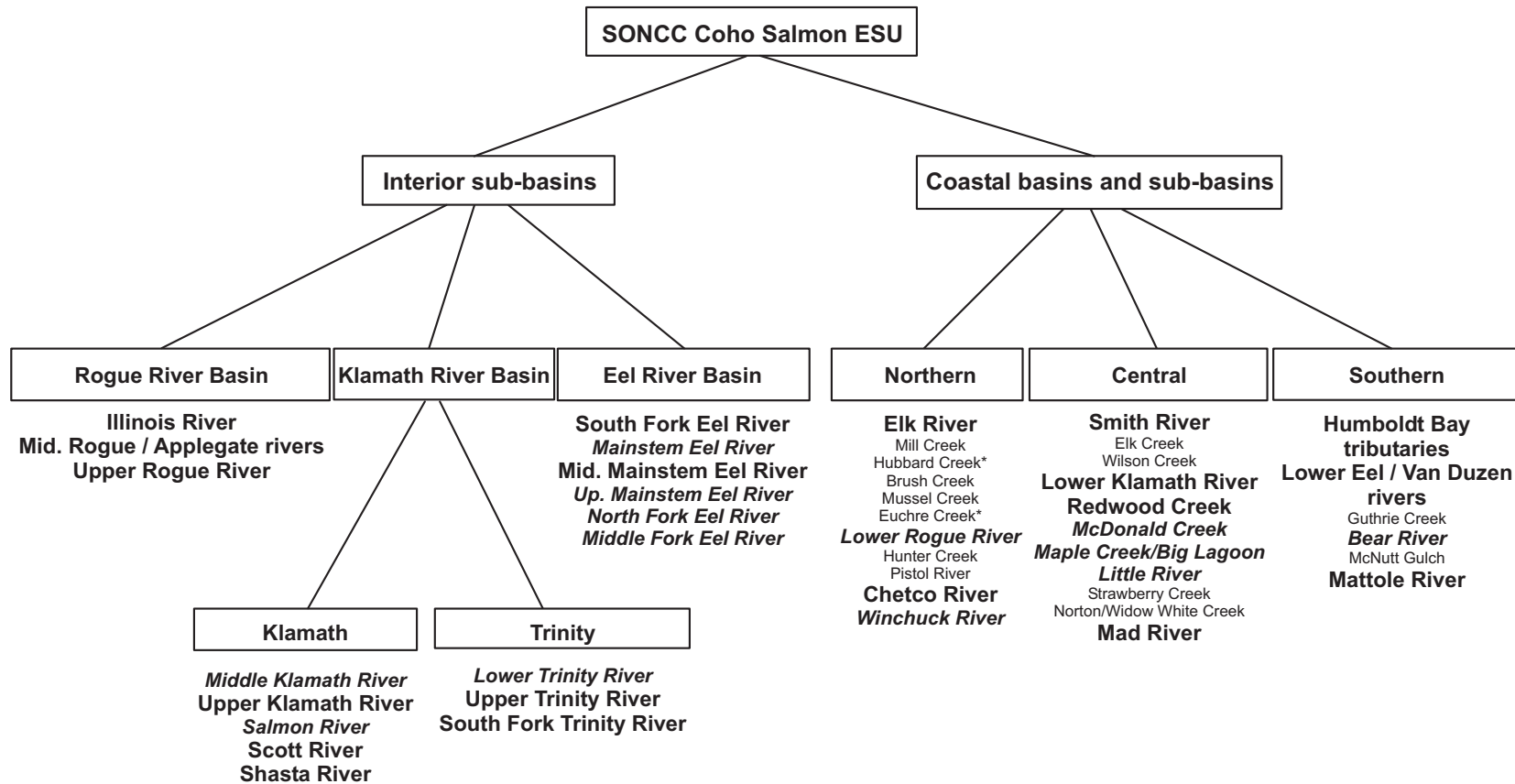


Figure 2. Arrangement of historical populations of the Southern Oregon/Northern California Coast Coho Salmon ESU into diversity strata. Functionally independent populations are listed in bold font, potentially independent populations are listed in bold italic font, other listed populations are dependent and ephemeral population

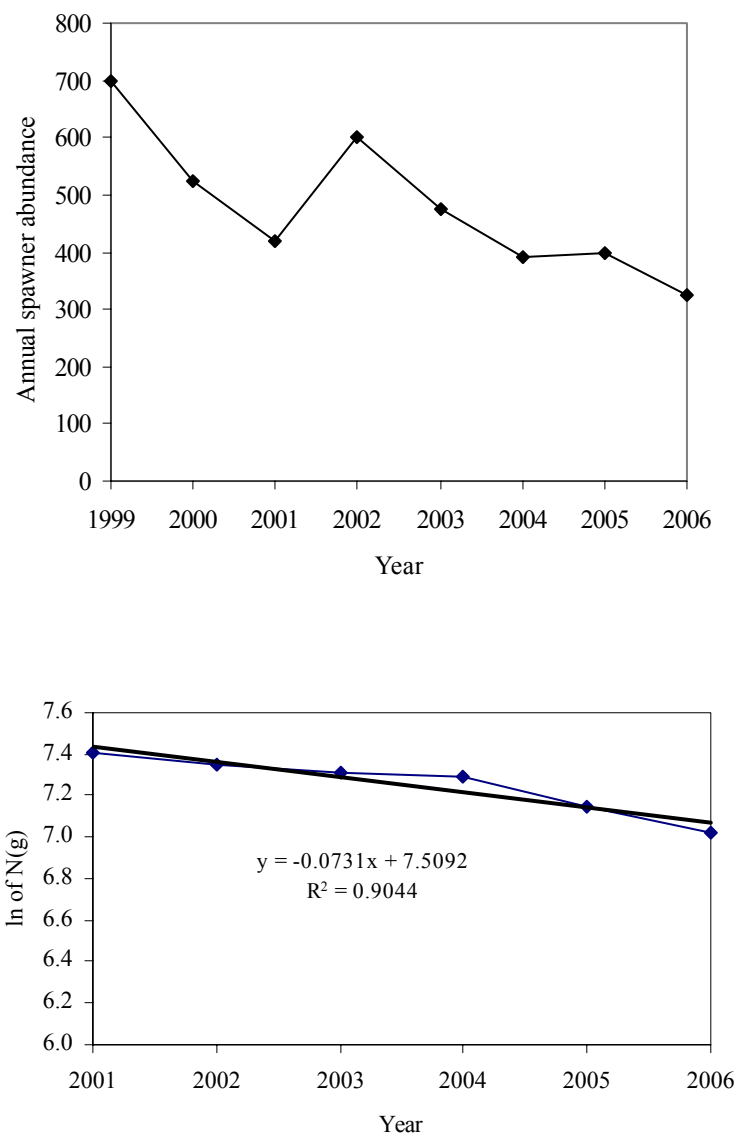


Figure 3. Annual spawner abundance over past eight years (upper plot) and calculation of population trend (lower plot) based on example data set in Table 2. The lower plot shows the regression of generational running sum (ln-transformed) and time.

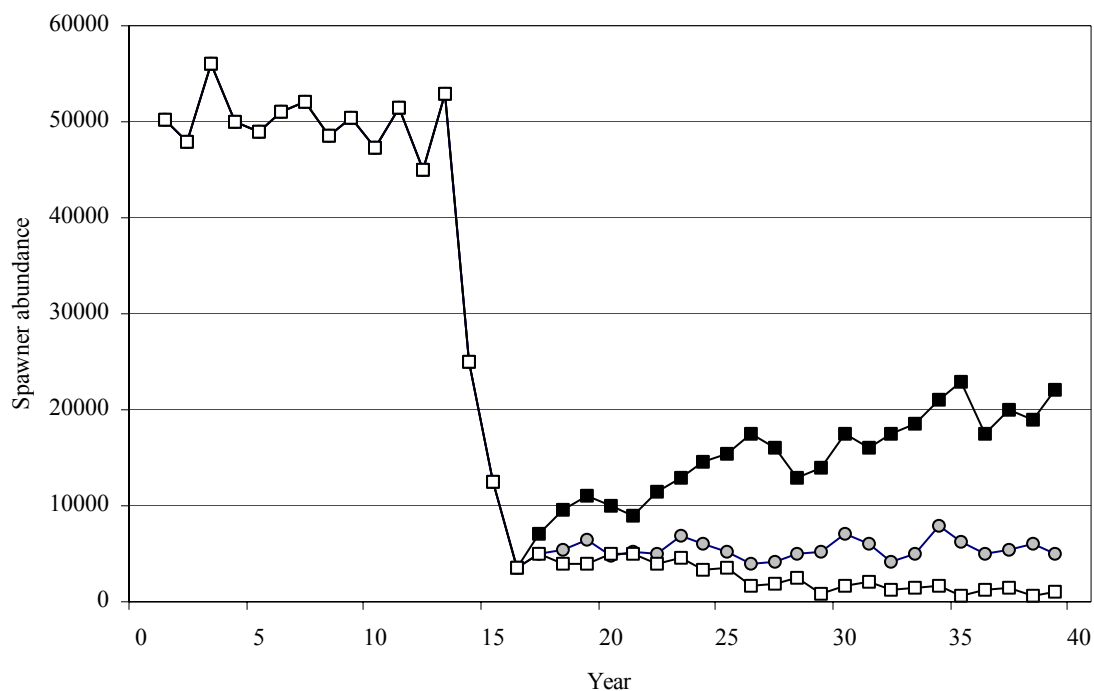


Figure 4. Hypothetical example of catastrophic decline in abundance showing three possible trajectories: a continued downward trend in abundance (open squares), a relatively stable abundance following decline (gray squares), and an apparent upward trend toward recovery following the decline (solid squares). Figure based on Spence et al. (In prep.).

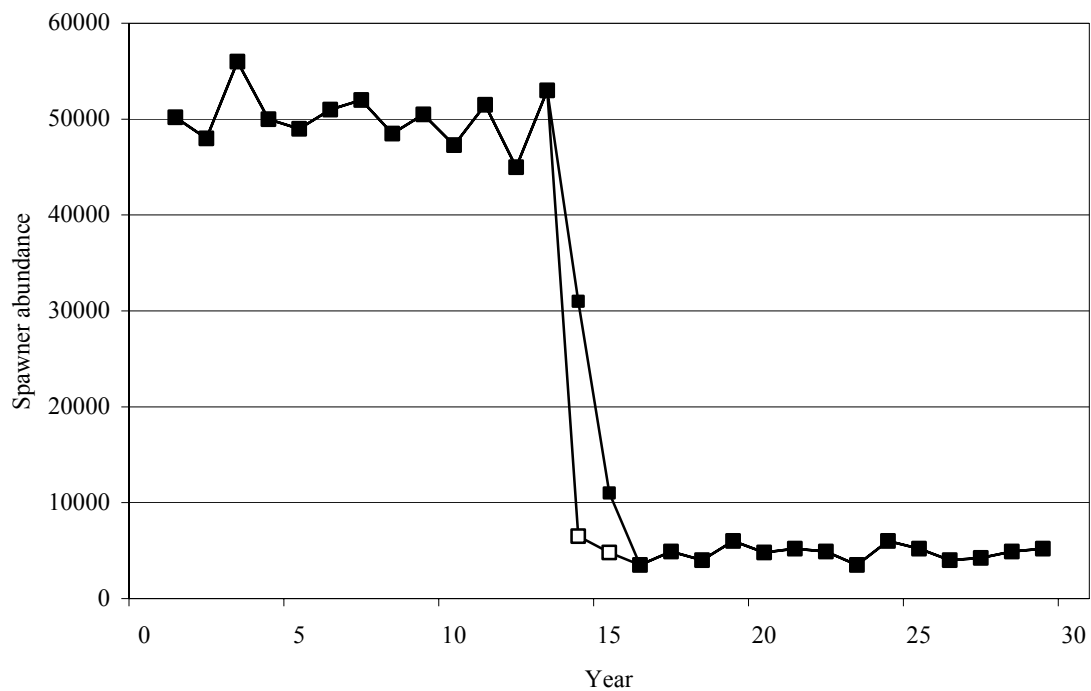


Figure 5. A hypothetical example where an order of magnitude decline in abundance occurs over a single year (open squares) versus three years (solid squares). Figure based on Spence et al. (In prep.).

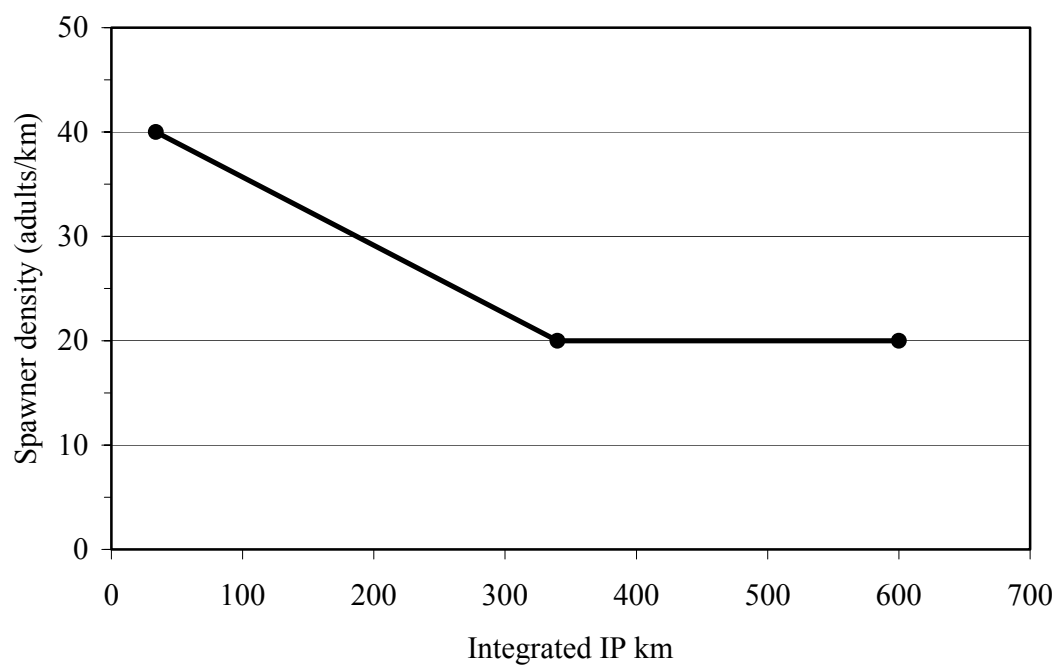


Figure 6. Minimum required spawning density (MRSD) based on amount of integrated IP-km for coho salmon.

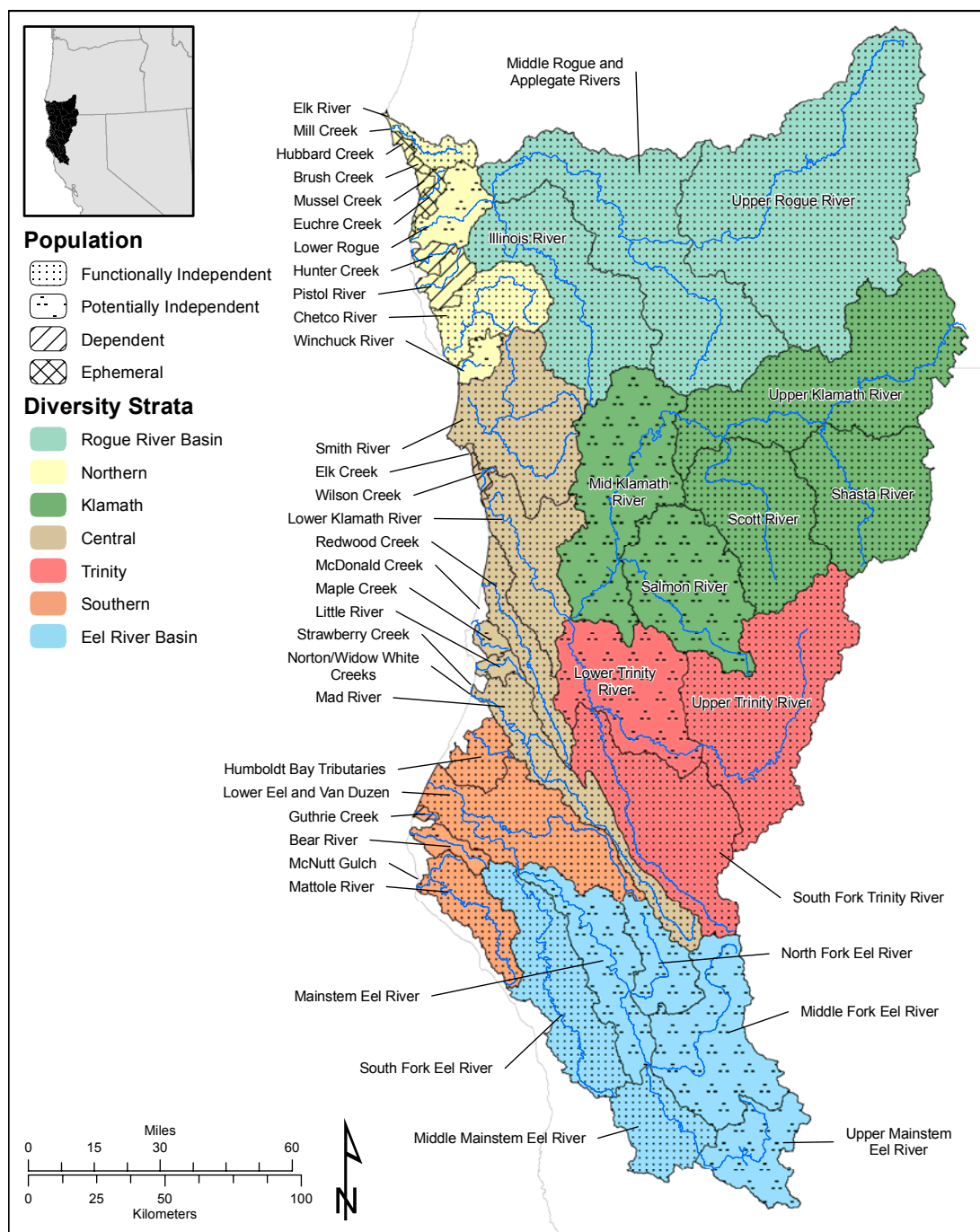


Plate 1. Diversity strata for populations of coho salmon in the SONCC ESU. Based on Williams et al. (2006).

